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Better air for better health: Forging synergies in policies for energy access, climate change and air pollution

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ABSTRACT

Air pollution and its related health impacts are a global concern. This paper addresses how current policies on air pollution, climate change and access to clean cooking fuels can effectively reduce both outdoor and household air pollution and improve human health. A state of the art modeling framework is used that combines an integrated assessment model and an atmospheric model to estimate the spatial extent and distribution of outdoor air pollution exposures. Estimates of household energy access and use are modeled by accounting for heterogeneous household energy choices and affordability constraints for rural and urban populations spanning the entire income distribution. Results are presented for 2030 for a set of policy scenarios on air pollution, climate change and energy access and include spatially explicit emissions of air pollutants; ambient concentrations of PM2.5; and health impacts in terms of disability adjusted life years (DALYs) from both ambient and household air pollution. The results stress the importance of enforcing current worldwide air quality legislation in addressing the impacts of outdoor air pollution. A combination of stringent policies on outdoor air pollution, climate change and access to clean cooking fuels is found to be effective in achieving reductions in average ambient PM2.5 exposures to below World Health Organization recommended levels for a majority of the world's population and results in a significant decline in the global burden of disease from both outdoor and household air pollution.

1. Introduction

Adverse health effects of air pollution (both outdoor (ambient) and household related) have drawn considerable attention over recent years with increasing epidemiological evidence for cardiovascular, asthmatic and other health related outcomes (Dockery et al., 1993; Pope et al., 2002). In spite of legislated air pollution policies in many countries, recent studies estimate that 80% of the world's population continue to be exposed to ambient pollution that far exceeds the WHO recommended Air Quality Guideline (AQG) of 10 $\mu g/m^3$ for long-term PM2.5 concentration levels (particulate matter with aerodynamic diameter smaller than 2.5 μm) (Van Donkelaar et al., 2010; Rao et al., 2012; Brauer et al., 2012). In addition, while evidence of the high pollutant emissions and exposures resulting from the poor combustion efficiency of traditional biomass systems is well established as a major contributor to household (indoor) air pollution in developing countries (Smith and Haigler, 2008), recent studies also indicate potentially significant implications for ambient air quality (Zhang et al., 2000). More recent estimates indicate that outdoor and household air pollution are globally among the leading causes of mortality and morbidity related outcomes (Lim et al., 2012).

As a policy response to growing concern over air pollution, OECD countries have already implemented stringent air quality controls for ambient air quality and many large developing countries are increasingly following suit (Klimont et al., 2013). Economic growth has also led to an improvement in the quality of available fuels and technologies in developing countries (Stern, 2006; Dasgupta et al., 2001; Smith et al., 2005). However, emissions from cooking stoves continue to be a major component of global anthropogenic particulate matter (e.g., (UNEP/WMO, 2011)) in particular in developing countries, for e.g., in Africa and South Asia where emissions from cooking stoves are well over 50% of anthropogenic sources (Bond et al., 2004a, 2013). Improved access to modern energy services including cleaner-combusting and more efficient cooking fuels like LPG, biogas, natural gas and

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advanced biomass stoves for developing country households is also on the policy agenda of many countries and has received an impetus through the newly launched initiative of the Secretary-General of the United Nations toward "Sustainable Energy for All" (http://www.sustainableenergyforall.org/) and the Global Alliance for Clean Cookstoves (http://www.cleancookstoves.org/). Several measurement campaigns have evaluated the performance of improved stoves and fuels, including the evaluation of climate relevant species (e.g., (Maccarty et al., 2007, 2010)), and the potential health benefits of their introduction (e.g., (Anenberg et al., 2012)). In addition to resulting in significant health benefits, recent assessments suggest that such residential cooking fuel and stove switching, may also have a greater potential to curb global warming by reducing black carbon emissions (Bond et al., 2013; Shindell et al., 2012). Climate change and energy efficiency related policies are additionally being undertaken in many countries and these are likely to cause energy transformations that will impact air pollution and health related outcomes in the future.

A number of recent studies have focused on co-benefits of reducing short-lived aerosols and the associated reduction in climate and health related impacts (see for example (Anenberg et al., 2010; UNEP/WMO, 2011; Shindell et al., 2012). There is also a growing body of research focusing on the public health and potential climate co-benefits of improving access to modern cooking fuels and stoves in developing countries (Bond et al., 2004b; Haines, 2007; Smith and Balakrishnan, 2009). This new scientific research is resulting in increasing public attention on these issues and pressure to forge synergies in these traditionally separate policy domains. Also highlighted by current research is the limited assessment of policy impacts and potential co-benefits, lack of integration of the short-term benefits of related policies, and a growing need for integrated analysis that combines sophisticated modeling of policies, behavior of regulated entities, atmospheric transport chemistry, climate science and health effects (see (Jack and Kinney, 2010; Bell et al., 2008) for discussion).

In this context, we examine scenarios of outdoor and household air pollution and related health impacts in 2030, given different sets of policies on air pollution, climate change and energy access which are presented in detail in the recently published Global Energy Assessment (Riahi et al., 2012). The specific goal of this paper is to assess how effective such policy combinations could be in delivering improved air quality and health related outcomes. The climate change related outcomes of these scenarios in terms of long-term radiative forcing and associated temperature change are presented in detail in (Riahi et al., 2012) and (Mccollum et al., 2013) and are not discussed here. We do not examine here the direct impacts of climate change on human health although this is an important area of research. The underlying modeling framework used in this paper has been presented in detail in (Rao et al., 2012) and combines an integrated assessment model and an atmospheric chemistry transport model for the spatial distribution of outdoor air pollution exposures globally. We explicitly model future household energy access and use by accounting for heterogeneous household energy choices and affordability constraints for rural and urban populations spanning the entire income distribution based on (Pachauri et al., 2013). We use WHO Comparative Risk Assessment methods (Ezzati et al., 2004) and include a number of updates to methodology based on recent literature to estimate both ambient and household health related outcomes of the chosen policies. Global results are presented for 2030 and include spatially detailed emissions of air pollutants, ambient concentrations of PM2.5, health impacts in terms of disability adjusted life years (DALYs) from both ambient and household air pollution, and the associated costs of policies.

2. Materials and methods

We use the IIASA integrated modeling framework similar to (Riahi et al., 2011), including the Model for Energy Supply Strategy Alternatives and their General Environmental Impact (MESSAGE) (Messner and Strubegger, 1995; Rao and Riahi, 2006; Riahi et al., 2007) for deriving global scenarios of air pollutants. Sectors included are power plants, industry (combustion and process), road transport, households, waste, agriculture, and large-scale biomass burning. Estimates of a number of GHGs and air pollutants including methane (CH_4), sulfur dioxide (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO), volatile organic compounds (VOCs), black carbon (BC) and organic carbon (OC) are derived from the MESSAGE model and further spatially detailed at a 1×1 degree resolution. We use inventory data described in (Granier et al., 2011) and an exposure-driven algorithm for the downscaling of the regional air-pollutant emissions projections based on (Riahi et al., 2011). We include in the MESSAGE model, a detailed representation of a number of air pollution policies and costs of such policies until 2030 (methodology described in (Riahi et al., 2011, 2012). We estimate global PM2.5 emissions based on the black and organic carbon emissions in the MESSAGE model, and include in addition, non-carbonaceous components from fly-ash; production; and building sources (see (Rao et al., 2012) for details).

Atmospheric concentrations of particulate matter, in particular PM2.5, are calculated with the off-line global TM5 chemistry-transport model (Dentener et al., 2005, 2006a,b; Bergamaschi et al., 2007, 2009; Fiore et al., 2009). The models and methodology and validation of the results for 2005 are explained in detail in (Rao et al., 2012; Brauer et al., 2012) and references therein. In short, TM5 uses a set of nested grids; with a state-of-the art 1×1 resolution over the continental source regions. TM5 calculates emissions of natural origin, and uses a set of gridded emissions of primary and secondary aerosols from the MESSAGE model. Gas phase chemistry is calculated in the TM5 model using a modified CMB4 (Carbon Bond Mechanism 4) mechanism, and used to compute the formation of sulfate and nitrate, which are assumed to be in thermodynamic equilibrium following the EQSAM2 (Equilibrium Simplified Aerosol Model version 2) module. Secondary organic aerosol formation is parameterized using the AEROCOM recommendations in (Dentener et al., 2006a). Using parameterizations of wet and dry removal based on (Huijnen et al., 2010), output generated for this publication consists of primary (e.g. black carbon and organic carbon) and secondary (e.g. SO_4, NH_4 and NO_3) aerosol concentrations. As outlined in (Brauer et al., 2012) and (Rao et al., 2012), aerosol concentrations in urban regions are likely to be elevated compared to rural regions within a 1×1 degree TM5 grid cell. To compute aerosol concentrations relevant for exposure of populations, an urban increment is calculated on the basis of the contribution of primary particulate matter emissions from transport, energy and industry, and high resolution information on the fraction of population living in urban regions and the underlying land area. This is similar to the approach used in other recent studies (for e.g., (Brauer et al., 2012; Rao et al., 2012)) and includes a representation of both urban and rural exposures (thus also representing effects of industrial sources and other hot spots typically located outside urban areas) in assessing total PM2.5 concentrations and related health impacts on a global scale.

Health impacts from outdoor air pollution in terms of disability adjusted life years (DALYs) are further estimated using methodology detailed in (Rao et al., 2012). We use WHO baseline scenario data (WHO, 2008) on DALYs until 2030 based on a 5% discount rate. We limit the analysis to adults over 30 years of age and use a concentration threshold range of 7.5–50 $\mu g/m^3$ for PM2.5 in this study based on (Cohen et al., 2005) and (Krewski et al., 2009) although there is now evidence of health impacts at also higher concentrations (see (Lim et al., 2012). Central estimates of cause-specific relative risk rates (RR) were based on (Cohen et al., 2005) and are as applied in (Krewski et al., 2009) (see Appendix B for definition and summary of risk rates used in this paper). We do not estimate the health related impacts of ozone, as it is generally assumed that they are an order of magnitude smaller than those of particulate matter. We exclude here the impacts of outdoor air pollution on children although there is recent evidence of increased acute lower respiratory infection (ALRI) from outdoor air pollution in children below the age of 5 years (Lim et al., 2012). We also note here that the outdoor and household related health impacts are not independent, i.e. we do not adjust for the share of outdoor air pollution that can be attributed to household burning of solid fuels. While recent evidence indicates that this fraction is quite significant in many developing countries and is estimated at around 16% globally (Lim et al., 2012), we do not correct for this due to inherent uncertainties in estimation. There is also recent literature which suggests that the composition of PM2.5 could potentially have implications for the impacts on health (see for example (Ostro et al., 2006, 2009) but we do not examine this here. Appendix C summarizes the methodology for estimating health impacts.

We use one of the core WHO health and development indicators, "proportion of population using solid fuels," as a proxy for actual exposure to household air pollution. We interpret this as those populations relying on all solid fuels including coal, charcoal, wood, crop, or other agricultural waste, dung, shrubs, grass, and straw. Future scenarios regarding household dependence on solid fuels are assessed using the MESSAGE-Access modeling framework (Riahi et al., 2012; Pachauri et al., 2013). Estimates of relative risks for disease due to household air pollution from (Desai et al., 2004; Wilkinson et al., 2009) are used to estimate the burden of those diseases with strong epidemiological evidence for an enhanced risk due to solid fuel use (see also Appendix B for definition and summary of risk rates). Similar to outdoor air pollution, we use mortality and morbidity related outcomes for adults over the age of 30 years but additionally include children below the age of 5 years, given long-standing evidence that household air pollution from solid fuel use is a leading risk factor for ALRI in young children (Smith et al., 2004; Lim et al., 2012). We estimate health effects by combining information on the exposed population and the fraction of current disease levels attributable to solid fuel dependence, using the methodology described in (Rao et al., 2012) and summarized in Appendix C.

We use a baseline scenario of global energy and GHG emissions described in detail in (Riahi et al., 2012) as the underlying basis for estimation of pollution related emissions. The scenario includes detailed estimates of solid fuel use in the household sector based on (Pachauri et al., 2013). This scenario combines assumptions about high population and relatively slow income growth with modest rates of technological change and energy intensity improvements, leading in the long term to high energy demand (global energy demand is projected to increase to 460 Exajoules (EJ) in 2030) and a long-term climatic response similar to the RCP 8.5 described in (Riahi et al., 2011). This scenario thus provides a good basis for estimating the impacts of stringent transformational policies on the energy system and related impacts in pollutant emissions.

In this paper, we focus on selected combinations of climate change, energy access and air pollution policies as shown in Table 1 to specifically examine the impacts of such scenarios on outdoor and household air pollution levels and related health impacts in 2030. A more in-depth description of such policies is provided in (Riahi et al., 2012). Appendix A describes in detail the air pollution control measures assumed.

Table 1
Policy scenarios.

Scenario	Air pollution	Climate change	Energy access*
a	No improvement in air quality legislations beyond 2005	No policies. Annual energy intensity (EJ/GDP) growth of 3% per year. 72 Gt CO_2 eq GHG emissions in 2030	No policies.
b	All current and planned air quality legislations until 2030 (fuel standards, emission limits, technology standards). (See Supplementary Online Material for detail.)	Same as "a"	Same as "a"
c	All current and planned air quality legislations until 2030	Limit on global temperature change to 2 °C in 2100 based on the Conference of Parties agreement in Copenhagen. Annual energy intensity reduction of 2.6% until 2050. Replacement of fossil-fuels; and increase in zero-carbon electricity production). 42 Gt CO_2 eq GHG emissions in 2030.	Access to modern cooking fuels for 0.5 billion additional people in Sub-Saharan Africa, Pacific and South Asia by 2030 compared to a no new access policies scenario. This would be achieved by means of fuel subsidies on modern fuels and microfinance to households for new stove purchases.
d	Stringent air quality legislations (end-of pipe controls and fuel improvements) corresponding to 70% of maximum technologically feasible reduction levels. (See Appendix B.)	Same as "c"	Same as "c"
e	Same as "d"	Same as "c"	Aspirational scenario reflecting universal availability of cooking fuels in Pacific & South Asia, sub-Saharan Africa based on the United Nations (UN) call for universal access to modern energy services by 2030. This would be achieved by means of fuel subsidies on modern fuels and microfinance to households for new stove purchases. All households switch to either cleaner combusting fuels such as LPG or biogas or advanced biomass stoves with emissions and efficiency characteristics similar to cooking with LPG.

* The current framework does not consider specific access policies in Centrally Planned Asia (CPA) and Latin America (LAM) where use of coal and biomass is prevalent in the household sector. However economic growth is assumed to lead to a decline in such fuel use. We also do not consider here the impacts of electrification but recent analysis indicates that the GHG and pollutant emissions of such a policy are not likely to be significant (see (Riahi et al., 2012)).

3. Results

3.1. Emissions and concentrations

Fig. 1 shows population (>30 years) weighted ambient concentrations of PM2.5 (including an urban increment factor, excluding dust, secondary organic aerosols and sea salt) in 2005 and 2030 for scenarios (a–e). This represents the average PM2.5 concentrations derived by weighting each underlying grid cell value by the underlying population over 30 years of age. Also shown (as insets) are global anthropogenic emissions of SO_2, NO_x, and PM2.5. We compare the estimated PM2.5 concentrations to the WHO AQG of 10 $\mu g/m^3$ for long-term PM2.5 concentrations and WHO Tier I–III levels of 10–35 $\mu g/m^3$ still associated with significant health risks but shown to be achievable with successive and sustained abatement measures (WHO, 2006).

Assuming no additional air quality legislation beyond those committed globally by 2005 as in scenario (a), is seen to lead to an overall increase in both pollutant emissions and PM2.5 concentrations of more than 50% by 2030 compared to 2005 levels, thus indicating the key role of air quality policies to control growing outdoor air pollution in the future. Assuming a full implementation of all currently planned air quality legislation in the 2010–2030 period as in scenario (b) is found to lead to a slight decline in global SO_2 and PM2.5 emissions of around 2% while NO_x emissions increase by 15% compared to 2005 levels. Global population-weighted anthropogenic PM2.5 concentrations in 2030 are estimated to rise to 34 $\mu g/m^3$ compared to 26 $\mu g/m^3$ in 2005. The globally modest impacts of currently legislated air quality policies are similar to findings in (IEA, 2011), and can be explained in particular by increased NO_x and PM2.5 emissions from the transportation sector in developing countries, which offset the reductions resulting from the implementation of air pollution policies in OECD countries. Additionally, there is an increase in the population dependent on solid fuels for cooking, which grows from 2.2 billion to 2.4 billion in South Asia, Pacific Asia and sub-Saharan Africa between 2005 and 2030, in the absence of new policies to improve access to clean cooking fuels. These trends are similar to those in (IEA, 2011) that estimate an initial increase in the number of people dependent on biomass fuels globally, and despite a drop in the share of population dependent on solid fuels, almost no change in the numbers by 2030.

The implementation of global climate mitigation policy and a partial energy access policy in scenario (c) lead to reductions of outdoor air pollution related emissions of SO_2, NO_x and PM2.5 of around 40% compared to scenario (b). The reductions derive mainly from resulting improvements in energy efficiency; substitution of fossil fuels; adoption of advanced energy technologies; and an overall increase in the share of zero-carbon electricity. Additionally, the population dependent on solid fuels is reduced by 0.5 billion in 2030 as a result of energy access policies (see Table 1 for details). The associated population-weighted global PM2.5 concentrations are estimated at 26 $\mu g/m^3$ in 2030. However, large fractions of the population are still exposed to ambient concentrations of PM2.5 above WHO recommended levels in many parts of the world. Also, around 1.9 billion people still remain dependent on solid fuels in 2030, and related cooking emissions continue to contribute significantly to both ambient and household pollution. It is necessary to note that we exclude here the contributions from dust and sea salt, which are significant in many parts of the world.

Increasing further the stringency of outdoor air pollution policies as in scenario (d), yields significant emission reductions compared to scenario (c), particularly in industrial and transportation sectors. As climate change and energy access policies are the same as in scenario c, the additional reductions in emissions derive primarily through a number of additional pollution controls on these sectors (see Appendix A for details). While in OECD regions, this results in up to 80% of the population below the WHO AQG level for PM2.5, in Asia and Africa, concentration levels remain high. A major contributor is the continued use of biomass for cooking in these regions, as a result of which household related PM2.5 emissions are high (40–50% of total PM2.5 emissions in 2030). Further inclusion of an energy access policy that eliminates the use of solid-fuels in the household sector as in scenario (e), is thus particularly effective in regions with high solid fuel use, and

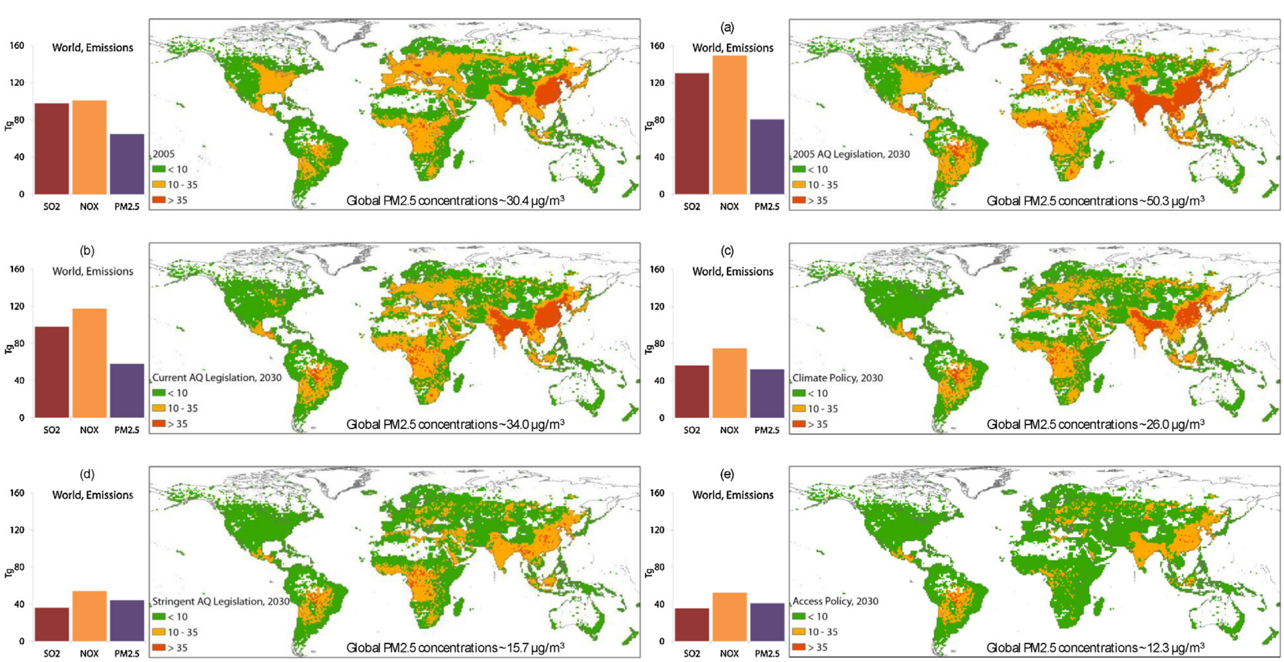

Fig. 1. Global population-weighted average anthropogenic PM2.5 concentrations (excluding dust, sea salt and secondary organic aerosols, including large scale biomass burning), and emissions of SO_2, NO_x and PM2.5 in 2005 and across scenarios in 2030 (a–e). The ranges for concentrations reflect the WHO recommended AQG value (10 $\mu g/m^3$), the attainment of which is expected to significantly reduce the health risks and the three interim targets defined for long-term PM2.5 concentration (10–35 $\mu g/m^3$).

results in 100% of the global population below 35 μg/m^3, and more than 50% of the global population at levels below 10 μg/m^3.

Our results thus lend further support to current evidence that currently legislated outdoor air pollution policies, while necessary to control growing air pollution, will be inadequate in meeting WHO recommended levels for long-term PM2.5 globally in 2030. Stringent pollution policies are found to be necessary in this context and can lead to significant improvements in global air quality when supplemented by policies on climate change and energy access.

3.2. Health impacts

Fig. 2 summarizes the health impacts for 2005 and in 2030 for the different policy packages (see also (Rao et al., 2012) for discussion of 2005 impacts). For 2005, we estimate 23 million DALYs from outdoor air pollution and 46 million DALYs from household air pollution. This can be compared to the most recent estimates from (Lim et al., 2012) of 76 million DALYs from outdoor air pollution and 116 million DALYs from household air pollution in 2010. Differences in both sets of estimates can be attributed to a number of reasons. While a difference in base-years is one component, inclusion of more disease categories in (Lim et al., 2012) compared to our study is another factor. These include in particular ALRI in children less than 5 years also for outdoor air pollution and cerebrovascular disease and lung cancer from biomass use as risk categories for household air pollution. Another important reason for differences is that (Lim et al., 2012) postulates non-linear concentration response functions and assumes continued risk of disease also at concentrations above 50 μg/m^3. Including these updates would affect our numbers, but there is unlikely to be changes in the relative health impacts of the scenarios.

If outdoor air pollution legislation remained frozen at 2005 levels (scenario a), this would lead to a global increase of nearly 50% in outdoor air pollution related DALYs in 2030 compared to 2005. Current outdoor air quality legislation (scenario b) results in lower health impacts as compared to scenario (a) in 2030, but DALYs still increase by more than 30% as compared to the 2005 level. While this is partly explained by the increase in PM2.5 concentrations, an additional factor for increased health impacts is the growth in population above 30 years of age, especially in developing countries, where large increases can be expected in these age cohorts in the next two decades. It is also necessary to

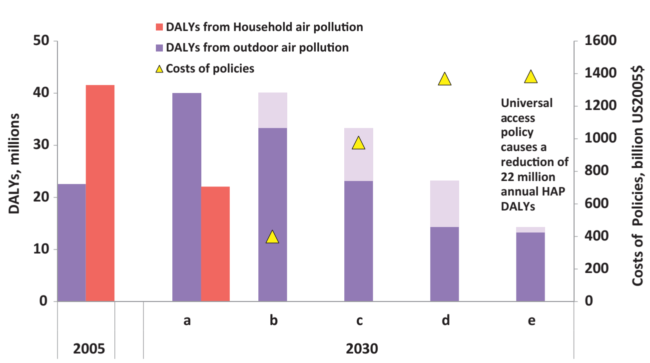

Fig. 2. Global Health Impacts (million DALYs) of outdoor and household related air pollution in 2005 and 2030 and Combined Annual Costs (US2005$), of air pollution, climate change and access policies in 2030. Household air pollution DALYs (red bars) only estimated for limited set of scenarios. Purple bars indicate impacts from outdoor air pollution. Shaded or lighter areas specifically indicate the savings in terms of outdoor air pollution related DALYs due to policies. Policy costs shown here on the right-hand axis are additional to annual energy system costs of US$1630 billion in 2030. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

note that the underlying share of cardiopulmonary disease and lung cancer related causes in the overall burden of diseases increases significantly from current levels in many developing countries in 2030, reflecting a baseline shift from infectious diseases to chronic ones, thus also contributing to the increased health impacts from outdoor air pollution. We further estimate that in the absence of any significant policies on energy access, 22 million DALYs can still be attributed to household air pollution in 2030. The reductions in household pollution related DALYs from 2005 levels accrue from a shift in underlying baseline mortality as DALYs attributable to ALRI among children are expected to decline between 2005 and 2030 even in the absence of any access policies (see (Riahi et al., 2012) for details).

Additional inclusion of climate change and partial energy access policies (scenario c) lead to a reduction of more than 7 million DALYs compared to scenario (b) as a result of lower PM2.5 concentrations due to the transformational changes in the energy system associated with these policies. Increased stringency of air pollution controls and universal energy access (scenario e), while eliminating the remaining household air pollution related DALYs (22 million DALYs saved from reduced household air pollution), also leads to a reduction of 20 million DALYs in outdoor air pollution impacts or more than a 50% reduction as compared to (c). The additional outdoor air pollution related health related impacts of the universal access policy, while small in global terms, are particularly significant for sub-Saharan Africa and South Asia, where as seen in Fig. 1, large declines in ambient concentrations are observed. The associated global burden of disease from outdoor air pollution reduces to less than 5% of the DALYs associated with cardiovascular, respiratory, and lung cancer related disease in 2030.

Our results provide further evidence that policies on climate change and energy access could potentially provide significant benefits in terms of outdoor air pollution related health outcomes. It is important to acknowledge that health impacts as estimated here are subject to a number of uncertainties and are used mainly to provide a comparative basis for the scenarios. Future work will need to take into account a number of recent updates in methodology and estimation as postulated by the new GBD study (Lim et al., 2012).

3.3. Costs

The costs of individual policies vary significantly based on the underlying type and stringency assumed. Air pollution policies result in total annual control costs in 2030 ranging from 400 billion US$ (scenario (b) up to 800 billion US$ per year (scenario (d)) (all economic values are expressed in 2005 US $). Climate change policies as in scenario (c) result in substantial additional energy system cumulative investments (including supply and demand) of 830 billion US$ per year in 2030. These estimates are similar to other recent studies (see for example (IEA, 2011). The costs of energy access policies (the investments needed for providing LPG or advanced biomass stoves to new users as well as the subsidy costs to lower prices of LPG) are estimated at between 3.5 billion US$ per year (partial access to modern cooking fuels as in scenario (d)) to 17 billion US$ per year (universal access to modern cooking fuels as in scenario (e)), most of which would be required in Sub-Saharan Africa and South Asia. Further details on the costs of individual policies are found in (Riahi et al., 2012).

Fig. 2 indicates the costs of the combined policy packages. We find that significant cost related co-benefits exist from viewing policies in combination. For instance, the direct costs of outdoor air pollution control are significantly reduced in scenarios (c) and (d) (up to 40% reduction in pollution control costs in 2030) due to the fuel shifts and efficiency improvements associated with climate change policies which limit the need for costly end- of- pipe controls. These findings are similar to other studies that also estimate a reduction in air quality control costs from the implementation of climate mitigation policies (see for example (Amann et al., 2011; Mccollum et al., 2013). Energy access policies are also seen to bring significant benefits in outdoor air pollution, at very little additional costs.

4. Discussion

Our results demonstrate the application of an extensive knowledge base and analytical tools that combine information on anthropogenic activities relating to air pollution and spatial dispersion of emissions, in order to examine how combined policies on air pollution, climate change and energy access could improve outdoor air quality and related health outcomes.

We underline the relevance of integration across multiple policy domains for improvements in outdoor air quality and the role that health related indicators can play in the evaluation and choice of such policies. We find that in many regions current air quality policies may be insufficient to achieve reductions in global air pollution. Beyond more stringent air pollution control, policies on climate change and energy access could promote important structural changes in the energy system which not only contribute to improvements in air quality in the shorter-term, but may also have potential impacts on outdoor air pollution levels and related health impacts in the longer-term. Energy access policies will also simultaneously lead to a corresponding decline in household air pollution and related health impacts. We find that a full suite of policies on air pollution, climate change and energy access results in more than 50% of the world's population below WHO recommended levels for long-term PM2.5 exposure in 2030.

Our results also provide support to the notion that the substantial and immediate health related benefits associated with climate change policies could potentially accelerate low carbon energy choices in the first place. The inclusion of outdoor air pollution related health benefits could further provide an additional impetus to energy access policies which are primarily focused on the issue of household air pollution.

We highlight the need for multi-criteria methodologies for evaluating the costs and benefits of combined policies, which could have significant impacts on the choice and evaluation of such frameworks. While we focus here on the avoided or reduced costs for pollution control, it is important to acknowledge that given the inherent difficulties in valuing human life in economic terms, the analysis does not attempt to quantify, for instance, the additional benefits associated with such policies, including for example reduced health expenditure. Further, we do not discuss the uncertainties in estimating the climate impacts of particulate matter reductions, which could significantly affect the evaluation of the co-benefits of such policies. It is also important to acknowledge that policy frameworks as discussed here presume the effective implementation of subsidies, regulation, enforcement and increased capacity building. The choice of policies, the stringency of the targets, and the exact combination of clean fuels and technologies will be specific to each country or region and are likely to be associated with significant uncertainties with regards to their design, costs and implementation. However, the analysis presented here clearly highlights the importance of potential air quality and health benefits from better coordination across policy domains that have traditionally been assessed in isolation from each other.

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Appendix A. Policies and measures for air pollution control

	Transport	Industry and power plants	International shipping	Other
Current legislation (CLE)				
Sulfur dioxide (SO_2)	OECD: EU fuel quality directive (2009/30/EC) and national legislation on the sulfur content in liquid fuels; non-OECD: national legislation on the sulfur content in liquid fuels and coal.	OECD: for EU, emission standards from the LCPD (2001), IED (2010), NEC (1991), UNECE (1999). National legislation elsewhere. Non-OECD: increased use of low-sulfur coal, increasing penetration of flue gas desulfurization (FGD) after 2005 in new and existing plants according to national legislation.	MARPOL Annex VI revisions from MEPC57.	Limiting open burning of agricultural waste (if legislation exists).
Nitrogen oxides (NO_x)	OECD: emission controls for vehicles and off-road sources up to the EURO-IV/EURO-V standard (vary by region). Non-OECD: National emission standards equivalent to approximately EURO III–IV standards (vary by region).	OECD: for EU, emission standards from the LCPD (2001), IEC (2010), NEC (1991), UNECE (1999). National legislation elsewhere. National emission standards on stationary sources – if stricter than in the LCPD. Non-OECD: primary measures for controlling of NO_x.	MARPOL Annex VI revisions from MEPC57.	Limiting open burning of agricultural waste (if legislation exists).
Carbon monoxide (CO)	As above for NO_x.			Limiting open burning of agricultural waste (if legislation exists).
Volatile organic compounds (VOC)	Measures as described above for NO_x; legislation on fuel quality and evaporative losses.	A number of directives for the EU: e.g., solvent directive of the EU (1999), stage I directive (1995), NEC (1991), UNECE (1999).		Limiting open burning of agricultural waste (if legislation exists).
Ammonia (NH_3)		End-of-pipe controls in industry (fertilizer manufacturing).		NEC (1991) and UNECE (1999).

Appendix A (*Continued*)

	Transport	Industry and power plants	International shipping	Other
PM2.5 (including BC and OC)	As for NO_x.	For the OECD like for SO_2, NOx; for the non-OECD, improving enforcement of PM control with end of pipe measures required by national legislation; often linked to FGD requirements.		Limiting open burning of agricultural waste (if legislation exists).
Additional measures in stringent legislation (SLE): corresponding to 70% of maximum technologically feasible reduction levels				
SO_2	As in CLE.	High-efficiency flue gases desulfurization (FGD) on existing and new large boilers. Use of low-sulfur fuels and simple FGD techniques for smaller combustion sectors. High-efficiency controls on process emission sources.	Same as CLE.	Reduction in agricultural waste burning.
NO_x	OECD and non-OECD: EURO-5 and EURO-6 for light duty vehicles.	Selective catalytic reduction at large plants in industry and in the power sector. Combustion modifications for smaller sources in industry and in the residential and commercial. sectors. High-efficiency controls on process emission sources.	Same as CLE.	Reduction in agricultural waste burning.
CO	As in CLE.			Reduction in agricultural waste burning.
VOC	As in CLE.	Regular monitoring, flaring, as well as control of the evaporative loses from storage. Solvent use: full use of potential for substitution with low-solvent products in both "do it yourself" and industrial applications, modification of application methods and introduction of solvent management plans.		Reduction in agricultural waste burning.
NH_3		End-of-pipe controls in industry (fertilizer manufacturing).		Substitution of urea fertilizers, rapid incorporation of solid manure, low nitrogen feed and bio-filtration.
PM2.5 (including BC and OC)	As in CLE.	High-efficiency electrostatic precipitators, fabric filters, new boiler types, filters, good practices.	Revised MARPOL Annex VI (2005) regulations.	Reduction in agricultural waste burning.

1. LCPD, 2001: Council Directive 88/609/EEC of 24 November 1988 on the limitation of emissions of certain pollutants into the air from large combustion plants.
2. IED, 2010: Industrial Emissions Directive (2010/75/EU) http://ec.europa.eu/environment/air/pollutants/stationary/ied/legislation.htm.
3. NEC, 1991: National Emission Ceiling Directive (2001/81/EC).
4. UNECE, 1999: The 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone; http://www.unece.org/env/lrtap/multi_h1.html.
5. Solvent Directive of the EU, 1999: 1999/13/EC Council Directive 1999/13/EC on the limitation of emissions of volatile organic compounds due to the use of organic solvents in certain activities and installations.
6. Stage 1 Directive, 1995: 1994/63/EC aims to prevent emissions to the atmosphere of volatile organic compounds (VOCs) from off-road sources.
7. MARPOL: International Convention for the Prevention of Pollution From Ships, 1973 as modified by the Protocol of 1978; Annex VI entered into force in 2005.

Appendix B. Risk rates[a] for outdoor and household air pollution

Relative risk rates for outdoor air pollution.

Health outcome	GBD category, WHO 2009	Group (sex, age in years)	Relative risk (per 10 $\mu g/m^3$)	Confidence Interval (CI)
Cardiopulmonary (infectious and chronic respiratory diseases and selected cardiovascular outcomes for adults.)	39,40,106–109, 111	Men and Women ≥ 30	1.059	1.015–1.105
Lung cancer	333	Men and Women ≥ 30	1.082	1.011–1.158

Relative risks for household air pollution from solid fuel use.

Health outcome	GBD category, WHO 2009	Group (sex, age in years)	Mean relative risk	Confidence interval (CI)
Acute lower respiratory infection (ALRI)	39	Children < 5	2.3	1.9–2.7
Chronic Obstructive Pulmonary Disease (COPD)	112	Women ≥ 30	3.2	2.3–4.8
Lung cancer (from exposure to coal smoke)	333	Women ≥ 30	1.9	1.1–3.5
Ischemic Heart Disease (IHD)	107	Women ≥ 30	1.2	n.a
Chronic Obstructive Pulmonary Disease (COPD)	112	Men ≥ 30	1.8	1.0–3.2
Lung cancer (from exposure to coal smoke)	333	Men ≥ 30	1.5	1.0–2.5

[a] The concept of risk rate (RR) relates to the relative risk of exposure that is associated with specific disease outcomes. In the case of household air pollution, the exposure is typically assumed to be based specifically on 24-h kitchen concentrations of PM2.5 typically in the range of 500–1000 $\mu g/m^3$, with 24-h personal exposures for cooks and young children around 200–500 $\mu g/m^3$, with short periods of peak exposure that are many times higher than these concentrations. In the case of outdoor air pollution, response functions for long-term exposure are based on 10 $\mu g/m^3$ increase in ambient PM2.5.

Appendix C. Methodology for estimation of health impacts from outdoor and household air pollution

We estimate health impacts from ambient air pollution using the population-attributable fraction (PAF) approach based on the gradient of risk between the theoretical minimum level of air pollution exposure and the estimated observed exposure (WHO, 2002). We apply an approach similar to that detailed in (Smith et al., 2004; Cohen et al., 2005) which involved: (1) estimating total population exposures to PM2.5; (2) choosing appropriate exposure-response factors for PM2.5; (3) determining the current rates of morbidity and mortality in the population of concern using data from (WHO, 2008) and (4) estimating the attributable number of deaths and diseases.

The population attributable fraction to exposure is calculated based on (Murray et al., 2003) and is estimated as:

$$\mathrm{PAF} = \frac{P*(RR-1)}{[P*(RR-1)+1]}$$

where P = exposure expressed in PM2.5 concentrations, and RR = relative risk for exposed versus non-exposed populations. Once the fraction of a disease that is attributed to a risk factor has been established, the attributed mortality or burden is the product of the total death or DALY estimates for the disease and the attributed fraction.

For household air pollution impacts, we estimate the effects by combining information on the exposed population and the fraction of current disease levels attributable to solid fuel use. The approach utilizes relative risk estimates for health outcomes that have been associated with exposures to household pollution due to indoor smoke from solid fuel use and uses the population dependent on solid fuels as an exposure surrogate. The fuel-based approach takes advantage of the large number of epidemiological investigations conducted primarily in rural areas of developed countries that treat exposure to household air pollution from solid fuel use (sfu) as a single category of exposure (Smith et al., 2004). Recent studies propose an integrated concentration-response curve that draws from studies on indoor air pollution, outdoor air pollution, and tobacco smoke (Lim et al., 2012) but we do not apply this here.

The attributable fraction (AF) to sfu, $\mathrm{AF_{sfu}}$, can be estimated as:

$$\mathrm{AF_{sfu}} = \left[\frac{p_e(r_{rr}-1)}{p_e(r_r-1)+1}\right] \tag{1}$$

where p_e represents the population exposed to the solid fuels and r_r the relative risk due to sfu.

Similarly, attributable burden due to the solid fuel, $\mathrm{AB_{sfu}}$ use can be estimated as

$$\mathrm{AB_{sfu}} = \mathrm{AF_{sfu}}\mathrm{CDL} = \left[\frac{p_e(r_r-1)}{p_e(r_r-1)+1}\right]\mathrm{CDL}$$

where CDL is the current disease level estimated from (WHO, 2008).

Appendix D. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.gloenvcha.2013.05.003.

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