Health Climate Change impacts report card technical paper

3. Climate Change Impacts on Human Health through its effect on Air Quality

Ruth Doherty[1], Mathew R. Heal[2], Fiona. M O' Connor[3].

1. School of GeoSciences, University of Edinburgh
2. School of Chemistry, University of Edinburgh
3. Met Office, Hadley Centre

Email: ruth.doherty@ed.ac.uk

Executive Summary

The World Health Organization has recently reported that evidence has strengthened for adverse health effects of the surface pollutants ozone (O_3) and particulate matter (PM). In Europe, emissions controls have resulted in declining trends of many emitted species. However, due to the complexities of the processes linking emissions and air quality, reductions in local or regional emissions do not always produce a corresponding fall in atmospheric concentrations. Hence, it is not possible to clearly attribute effects of emissions relative to effects of meteorology and climate on air quality, or to identify how health burdens have changed in recent decades, due to observable climate change. There is considerable literature, including the Intergovernmental Panel on Climate Change's (IPCC) 5th assessment report (AR5) that assigns confidence statements (as given below) to their findings, on the effects of climate change on O_3 but far fewer studies on the effects of climate change on PM concentrations. This leads to low confidence in the overall impact of climate change on $PM_{2.5}$. A number of studies have examined future climate and emissions changes and there is consensus that the impact of emission changes on air quality in the near-term will outweigh that due to climate change (medium confidence). Under the IPCC AR5 Representative Concentration Pathways (RCPs), O_3 averaged over Europe is expected to decrease under all scenarios due to higher water vapour concentrations, except under RCP8.5 when high methane (CH_4) emissions offset this decrease (high confidence). In polluted areas with high levels of nitrogen oxides (NO_x), elevated surface temperatures are likely to enhance chemical reaction rates and increase emissions, increasing mean surface O_3 (medium confidence).

Air pollution episodes are associated with stagnation events and sometimes heat waves. Air quality during the 2003 heatwave over Europe has been examined in numerous studies and mechanisms for enhancing O_3 have been identified. Anthropogenic climate change has increased the near-term risk of heat waves (high confidence). However, projections of changes in stagnation events associated with winter and summer air pollution episodes remain highly uncertain. A number of modelling studies suggest that uniformly elevated temperatures due to climate change will also increase peak O_3 and PM concentrations (medium confidence), but there is large variability in the patterns and incidences of these events across the studies.

Recent studies have sought to evaluate changes in future air pollution impacts on human health. However, besides considerable uncertainty because of the multitude of mechanisms by which changes in emissions and in climate can influence O_3 and PM concentrations there is also uncertainty in risk estimates to apply- including their spatial and temporal variations. For O_3, the unresolved issue of a threshold concentration below which no adverse effects occur is highly important for determining human health burdens.

1. Introduction and Scope

This review examines the current evidence for the effects of climate change on ambient air quality and on the consequent health impacts. The topics covered in this review are summarised in Figure 1.

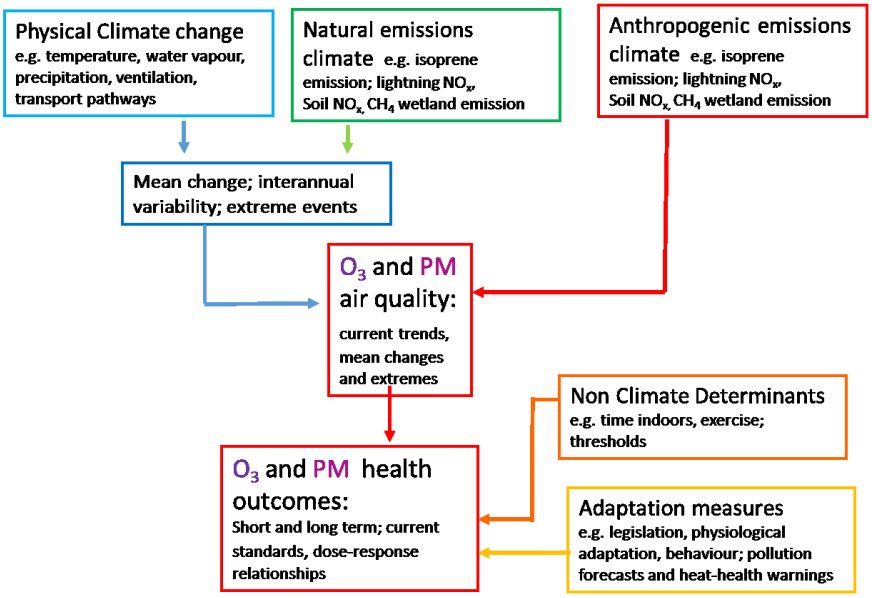

Figure 1: A summary of how climate change influences air quality and air quality-related human health including health modifiers and health adaptation measures.

First, the health impacts are outlined for the air pollutants with the greatest human health burden: nitrogen dioxide (NO_2), ozone (O_3) and particulate matter (PM). The sensitivity of these pollutants to the various meteorological and climate processes that affect their concentrations is then described. Air pollution during the 2003 heat wave is reviewed and key processes leading to poor air quality are identified. Current air pollution trends across Europe and in the UK are then described and current ranges of risk estimates summarised. Studies that have attempted to quantify global health burdens due to anthropogenic climate change are outlined and the difficulties in attributing health impacts to observable climate change highlighted. Future air quality projections and associated health impacts are then described. It is important to highlight that future air quality depends on many factors: anthropogenic emissions (global and regional), natural emissions (that are climate sensitive), as well as physical climate. Hence, three sets of studies are reviewed: studies that examine climate change (including natural emissions) impacts on air quality, studies that examine the impacts of future emission scenarios on air quality, and studies that examine the impacts of the combination of climate and emissions changes on air quality. Studies that relate these air quality impacts to health burdens are presented in the relevant sections. Future changes in extreme pollution events are also presented and key non-climate/emissions determinants of future air quality described. Potential adaption measures/issues as well as current forecasting strategies in the UK are outlined. Finally, this review presents conclusions and discusses knowledge gaps.

2. The health outcomes of air pollution

The World Health Organisation (WHO), amongst other international and national bodies, undertakes periodic reviews of the accumulating evidence from toxicological and epidemiological research on the detrimental effects of exposure to ambient air pollution on human health. Ambient air pollutants include particulate matter (PM), ozone (O_3), nitrogen dioxide (NO_2), sulphur dioxide (SO_2), and a number of toxic metals (e.g. Pb, Hg, Cr, As, Ni) and organic compounds (e.g. polycyclic aromatic hydrocarbons (PAHs) and dioxins). The greatest estimated health burdens globally, derive from exposure to PM, O_3 and NO_2 (WHO, 2006) and the most recent WHO review reports a strengthening of evidence for health impacts associated with exposure to these pollutants (WHO, 2013).

For PM, both short-term and long-term exposure to $PM_{2.5}$ (measured particle diameter <2.5 µm) is associated with all-cause and cardiovascular mortality and morbidity. Evidence now also links long-term exposure with other health outcomes including adverse birth outcomes, childhood respiratory disease, and possibly also with neurodevelopment, cognitive function and diabetes (WHO, 2013). There is also strengthening evidence for short-term effects on mortality and morbidity from the larger particle size fractions PM_{10} (diameter <10 µm) and PM_{coarse}, the latter being difference between the PM_{10} and $PM_{2.5}$ metrics.

For O_3, short-term exposure is associated with all-cause, cardiovascular and respiratory mortality, respiratory and cardiovascular hospital admissions and a range of other adverse pulmonary endpoints. Crucially, there is now strengthened (but not wholly consistent) evidence for effects of long-term exposure to O_3 on respiratory and cardiorespiratory mortality, on mortality effects for those with predisposing conditions such as Chronic Obstructive Pulmonary Disease (COPD), myocardial infarction and diabetes, and for adverse effects on asthma incidence, asthma severity, and lung function growth (WHO, 2013).

For NO_2, evidence for impacts of both short-term and long-term exposure on mortality, hospital admissions and respiratory symptoms, independent of the impacts of other air pollutants has strengthened, although, because of strong correlations between NO_2 and other air pollutants, it remains difficult to discern a direct effect (WHO, 2013).

For both PM and O_3 there are indications that adverse effects exist down to low concentrations, i.e. it is hard to discern any threshold concentration.

3. Sensitivities of air pollution to meteorology, extreme events, and to climate change

To understand the sensitivity of air pollution to climate, knowledge of atmospheric composition and the processes controlling it are crucial. NO_2 and O_3 are gaseous pollutants whilst PM comprises numerous aerosol species which are solids or liquid particles suspended in air. NO_2 is mainly formed from emissions of nitric oxide (NO) which is oxidised to produce NO_2. O_3 is formed from photolysis of NO_2. Volatile Organic Compounds (VOCs) including methane (CH_4) produce the highly reactive radical species that oxidise NO. PM is comprised of primary (directly emitted) and secondary (produced in the atmosphere) inorganic and organic species of varying sizes. Secondary inorganic PM (a major component of $PM_{2.5}$) comprises sulphate and nitrate aerosols formed from oxidation of SO_2 and NO and NO_2 (to HNO_3) and reactions with emissions of ammonia (NH_3). An important source of Secondary Organic Aerosol (SOA) is via the atmospheric oxidation of biogenic emissions of isoprene, a natural VOC. The pollutants are removed by dry deposition to the surface and, for PM in particular, by wet deposition in precipitation.

The lifetime of NO_2, considering conversion to HNO_3 and its subsequent removal as its only sink, is ~1 day (Moxim et al. 1996; Jacob, 1999). However if NO_2 is lofted high into the cold upper troposphere it can reside for several months in the form of peroxyacetylnitrate (PAN) until subsidence to warmer air re-releases NO_2 that can contribute to local pollution downwind (e.g. Real et al. 2007, 2010). On the other hand, O_3 has an average tropospheric lifetime of about a month (e.g. Stevenson et al. 2006; Young et al. 2013) enabling long-range transport around the northern hemisphere. The different components of PM have varying lifetimes but typical values are around ~1-2 weeks

(Jacobson, 2002), which is sufficient for transport across Europe as typical horizontal transport times across the mid-latitudes are ~2 weeks (Jacob, 1999).

The short-term (e.g. daily average) and long-term (e.g. annual) concentration of an air pollutant at a given location is determined by a large number of processes (emissions, transport, transformation and deposition), all of which can been affected directly or indirectly by meteorology and climate (the long-term average weather. The impacts of meteorological and climate variables on surface air pollution of O_3 and PM, are summarised in Figure 2. For further details, including the confidence in current understanding, see Isaken et al. (2009), Jacob and Winner (2009) and Fiore et al. (2012).

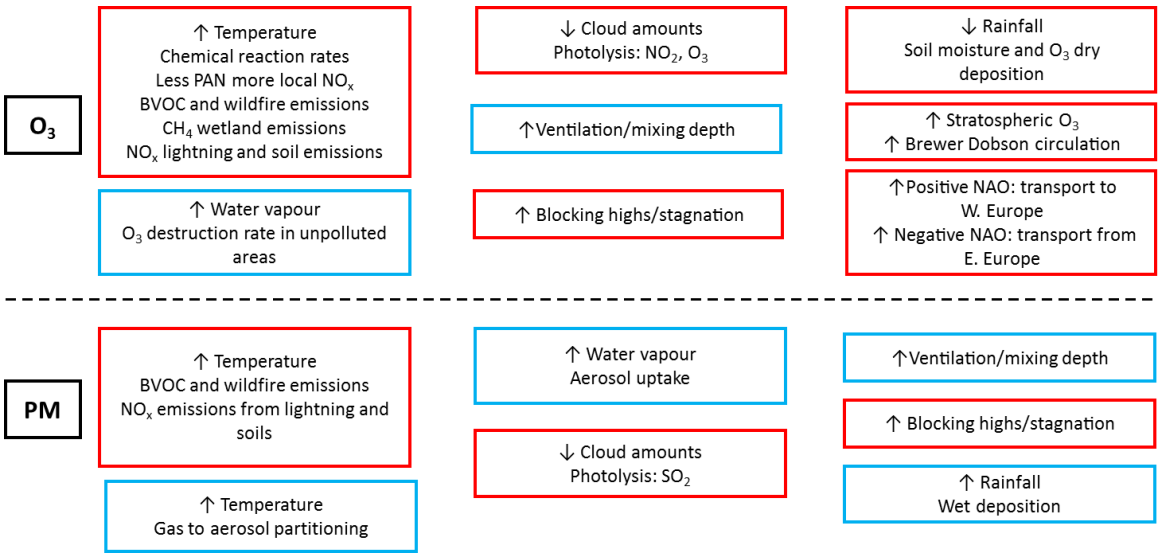

Figure 2: The impacts of meteorological and climate variables on O_3 and PM concentrations as determined from studies in the literature and adapted from Fiore et al. (2012) and references therein. ↑ represents an increase in the variable and↓ a decrease. Red (blue) box outlines depict where changes in a given variable predict an increase (decrease) in O_3 or PM.

Elevated temperatures increase chemical reaction rates leading to greater production and loss of O_3 and enhanced SO_2 oxidation to sulphate aerosol, but also leads to decreased partitioning of NH_3 and HNO_3 to nitrate aerosol. Natural emissions of isoprene strongly increase with increasing temperature (but also decrease with increasing atmospheric carbon dioxide CO_2). Sustained elevated temperatures can lead to reduced soil moisture, which decreases dry deposition of O_3 through plant stomata, and to increased wildfires which contribute to air pollutant emissions. Temperature also indirectly affects anthropogenic emissions through demands on energy consumption.

Rainfall frequency and amount affects wet deposition processes that remove pollutants from the atmosphere. Specific humidity also plays an important role in chemistry as higher water vapour increases O_3 destruction rates as well as altering aerosol properties. Cloud amount also affects the amount of incoming solar radiation and hence photolysis rates of NO_2 and of O_3.

Meteorological transport, in particular wind speed and direction, boundary layer height, vertical mixing, determines the dispersion, deposition or stagnation of pollutants and their precursors. Transport pathways themselves may be influenced by the dominance and passage of both low and high pressure systems, e.g. large-scale blocking over Europe may influence stagnation over the UK although there is no evidence of a direct relationship in this case (Webber et al. 2015 in prep).

In the context of sensitivity of air pollution to climate factors, this has been investigated in the context of: interannual climate variability, extreme events (such as heatwaves) and climate change, as discussed below.

Several studies using observations and models have examined how O_3 is modified by year-to-year variability in climate as regulated by the El Niño Southern Oscillation (ENSO) and the North Atlantic Oscillation (NAO) large-scale modes of climate variability. The NAO which represents the intensity of the pressure gradient and hence the airflow across the North Atlantic exerts an influence on the storminess of weather across the UK and the rest of Europe (e.g. Hurrell et al.1995). A number of studies have also examined the link between the NAO and O_3 over Europe. Li et al. (2002) found the influence of North American pollution on surface O_3 at Mace Head (Ireland) to be strongly correlated with the NAO, especially in spring. Creilson et al. (2003) also reported a relationship between the NAO and tropospheric O_3 over Western Europe in spring. Beside the NAO affecting pollution imported into Europe, several studies also find interannual variability in European pollution export to be strongly associated with the NAO (Eckhardt et al. 2003, Duncan and Bey, 2004). Two recent studies suggest that the positive NAO phase (strong pressure gradient) is associated with pollution transport from North America to northern Europe and the negative NAO phase (weak pressure gradient) is associated with weak transport of European pollution over western and central Europe (Christiados et al. 2012, Pausata et al. 2012). Overall, Pausata et al. (2012) reports that the NAO affect monthly surface O_3 concentrations by over 10 ppbv over Northern Europe and the British Isles in all seasons.

Another approach to study how annually-varying meteorology affects atmospheric composition of NO_2 and O_3 has been to perform model simulations with fixed and annually-varying meteorology or emissions (Uno et al. 207; Savage et al. 2008, Voulgarakis et al. 2010) or to use different meteorological datasets (Hess and Mahowald, 2009). In one such study, Voulgarakis et al. (2010) suggested that changing meteorology of winds, temperatures, humidity and clouds were important factors driving interannual variability of NO_2 and O_3 over a 5-year period.

Elevated air pollution associated with extreme events such as the summer heat waves has received much attention in the literature. In particular, the European heat wave of 2003 and the accompanying high O_3 levels has been studied in detail (e.g. Lee et al. 2006; Solberg et al., 2008, Vieno et al. 2010, Francis et al. 2011). Figure 3 shows that the 95-percentile of daily maximum hourly O_3 concentration in 2003 exceeded 160 µg m^{-3} over a large region in central Europe including southeast UK. These values exceeded the previous 95-percentiles for 1992-2002 for many measurement sites.

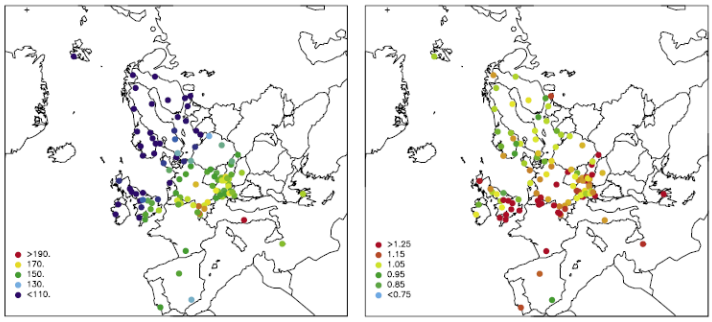

Figure 3. (left) The 95-percentile of daily maximum hourly O_3 concentrations observed in 2003 in □g m^{-3} and (right) this 95-percentile relative to the 95-percentile of the daily maximum hourly O_3 values observed during the period 1991–2002. Source: Solberg et al. (2008).

A number of studies have highlighted the role of the persistent high-pressure systems during this heatwave in trapping pollution (Vautard et al. 2005; Solberg et al. 2008) with subsequent downward entrainment of O_3-rich air to the surface over the UK (Francis et al.

2011). Solberg et al. (2008) discuss a number of further processes that contributed to elevated surface O_3 over N. Europe including: extended residence time of air parcels in the atmospheric boundary layer, extensive forest fires on the Iberian Peninsula, and enhanced levels of biogenic isoprene. Lee et al. (2006) showed that the elevated O_3 at Writtle, northeast of London (5 days with O_3 in excess of 220 µg m^{-3}) was due both to long-range transport of O_3-rich air from mainland Europe and to local O_3 production from elevated isoprene emission. Subsequent modelling studies also highlighted the potential role of suppressed dry deposition of O_3 due to stomatal closure in response to drought conditions (Vautard et al. 2005; Solberg et al. 2008; Vieno et al. 2010). In addition, Vieno et al. (2010) showed that for this event, over the UK, it was a multitude of factors operating on different days ranging from the impact of elevated temperatures on biogenic isoprene emissions, supressed dry deposition and transboundary import that contributed to the high levels of surface O_3. In a study of the effects of dry conditions on O_3 deposition during the heatwave in summer of 2006 across the UK, Emberson et al. (2013) reported that the number of days when daily maximum 8-hour average O_3 concentration exceeded 100 µg m^{-3} increased when the soil moisture drying effects were included.

Much less research has examined PM levels during the 2003 heatwave ($PM_{2.5}$ episodes tend to be more frequent in winter (Defra 2013); but as discussed earlier SOA formation can be enhanced with elevated isoprene emission during heatwaves). In one modelling study, wild fires were found to increase ground-level concentrations of $PM_{2.5}$ by ~10% over northern Europe (Hodzic et al.,2007).
All the above studies note the relevance of their results in relation to future changes in meteorology and climate as well as the increased likelihood of heat waves with climate change. Recent studies based on regional climate modelling suggest that summer 2003 could be a normal summer in the coming decades (Beniston, 2004; Schär et al.,2004).

Given that meteorology and climate are determinants of air quality, clearly any changes to climate (which will necessarily also include changes to meteorology) can affect air quality and consequently human health. Robust effects of mean climate change on O_3 include: increased temperatures leading to elevated water vapour concentrations leading to a decrease in 'background' O_3 (Jacob and Winner, 2009; Johnson et al. 1999; Doherty et al. 2013), and increased temperatures leading to faster chemical reaction rates in particular PAN decomposition enhancing surface O_3 concentrations in polluted areas (Sillman and Samson, 1995; Jacob and Winner, 2009; Doherty et al. 2013). Elevated temperatures have also been associated with increased BVOC emissions, increased O_3 and SOA formation (Racherla and Adams, 2008; Jacob and Winner, 2009). These processes are discussed in more detail in section 5. In addition, O_3 and some PM components are referred to as a short-lived climate forcers, as they also influence climate (e.g. O_3 is a greenhouse gas) on short timescales. However, a fuller discussion of short-lived climate forcers is outside the scope of this paper.

Meteorological and climate factors not only influence air pollutant concentration but they can also act as confounding and modifying factors in the air pollution concentration-health response relationship. An example of confounding is that high and low ambient temperatures have significant direct adverse health impacts, but yet also influence air pollution concentrations through the processes described above. An example of modification is where high (or low) temperatures make some parts of the population more susceptible to the adverse effects of air pollution (Pattenden et al., 2010); or where changes in weather or climate lead to lifestyle/behavioural changes that alter the exposure to air pollution.

4. Current air pollutant trends and health impacts attributable to observed climate change

4.1 Current air pollution trends and health burdens

The WHO publishes air quality guidelines for ambient air pollutants with widespread human health exposure (WHO, 2006). In the EU, two Air Quality Directives, 2008/50/EC and 2004/107/EC, set legally-binding limits and target values for a number of ambient air pollutants. The UK follows the EU legislation or in some cases sets more stringent values. Emissions of the main air pollutants in Europe have declined in the last decade or so, resulting for some pollutants in improved air quality across the region. However, due to the complexities of the processes linking emissions and air quality including interactions with meteorology, reductions in local or regional emissions do not always produce a reduction in atmospheric concentrations.

Table 1: Percentage of the EU urban population currently exposed to ambient air pollutant concentrations above EU and WHO reference levels. The range in estimates indicates the interannual variability in the 3-year period 2010-12. For pollutants subject to more than one air quality metric the most stringent to meet is used in this table. Where a limited number of exceedances is allowed for some metrics this has been taken into account. Source EEA (2014).

Pollutant	**EU limit or target value (averaging period)**	**% of urban population exposed to greater**	**WHO air quality guideline (averaging period)**	**% of urban population exposed to greater**
$PM_{2.5}$	25 µg m^{-3} (annual)	10–14	10 µg m^{-3} (annual)	91–93
PM_{10}	50 µg m^{-3} (daily)	21–30	20 µg m^{-3} (daily)	64–83
O_3	120 µg m^{-3} (max daily 8 hour)	14–17	100 µg m^{-3} or 50 ppbv (max daily 8 hour)	95–98
NO_2	40 µg m^{-3} (annual)	8–13	40 µg m^{-3} (annual)	8–13
SO_2	125 µg m^{-3} (daily)	< 1	20 µg m^{-3} (daily)	36–43

Table 1 shows that large proportions of the EU urban population remain exposed to ambient PM, O_3, NO_2 concentrations exceeding EU limit or target values, and even greater proportions are exposed to concentrations exceeding WHO air quality guidelines (AQG). The proportions have declined somewhat over the last 12 years for exposure to NO_2, but less so for exposure to PM, and are almost unchanged for exposure to O_3 (EEA, 2014).

It is estimated that in 2012, 21% of the urban population in the EU were living in areas where the 24-h limit value for PM_{10} was exceeded. For NO_2, it is estimated that 8% of the EU urban population in 2012 lived in areas where the annual EU limit value and the WHO AQG were exceeded. Although this proportion has declined, the decline is substantially less than the reduction in EU emissions of NO_x, attributed in part from the increased penetration of new diesel vehicles whose fraction emissions of NO_x as NO_2 is higher. For O_3, it is estimated that in 2012 14% of the EU urban population were living in areas exceeding the EU target value, but 98% were living in areas where O_3 exceeded the more stringent WHO AQG. In contrast to the situation for PM and NO_2, people living in rural areas are exposed to higher concentrations of O_3 than those in urban areas.

The UK is required to report annually to the EU on compliance with the EU directives (Defra, 2013). In 2012, all UK zones met the limit value for annual mean PM_{10}, and the

target value and Stage 1 limit value for $PM_{2.5}$ (25 µg m^{-3}) for 2015. The UK has an additional requirement for a 15% reduction between 2010 and 2020 in the 3-year average exposure indicator for $PM_{2.5}$ (a quasi-population-weighted urban background concentration) from its 2010-12 3-year average value of 13 µg m^{-3}. In 2012, there were widespread exceedances in the EU long-term objective for O_3 (in 41 out of 43 UK zones). There were also widespread exceedances in the EU limit value for annual mean NO_2 concentration (34 zones), although much of this exceedance is close to roads rather than generally across the urban background.

The estimated health burdens attributable to ambient air pollution are substantial. For O_3, the EEA estimated that short-term exposures in 2000 contributed to more than 20,000 premature mortalities (EEA, 2005). This assumes a threshold concentration for effect of 35 ppb. In the UK, it has been estimated that short-term exposures to O_3 in 2003 contributed to11,500 deaths brought forward and 30,700 respiratory hospital admissions, assuming exposure over the full year and no threshold for effects (HPA 2012). The estimated health burdens for the UK are 1,160 and 3,210, respectively, if a 35 ppb threshold is assumed (Heal et al., 2013). The estimated burden on the mortality of the UK population exposed to the anthropogenic $PM_{2.5}$ levels prevailing in 2008 for the rest of their lives is 340,000 life years lost (COMEAP, 2010). The burden can also be represented as an average loss of life expectancy from birth of 6 months (COMEAP, 2010). This compares with estimated average loss of life expectancies in the UK of 1-3 months from road traffic accidents and 2-3 months for exposure to passive smoking (EAC, 2010), although clearly in each case the actual loss of life expectancy will vary widely amongst individuals. Across the EU, the EEA estimate that about 350,000 people died prematurely in 2000 due to the ambient $PM_{2.5}$ which corresponds to a greater average loss of life expectancy of about 9 months compared with the average for the UK (EEA, 2005). Similar numbers have been derived for the USA; for example, a decrease of 10 □g m^{-3} in the concentration of $PM_{2.5}$ has also been shown to be associated with an increase in mean (□1 standard error) life expectancy of 0.61□0.20 year for populations in 211 counties within 51 US metropolitan areas (Pope et al., 2009).

Brauer et al. (2011) generated 0.1□ × 0.1□ spatially-resolved global estimates of long-term average ambient concentrations of $PM_{2.5}$ and O_3. Using these data, the Global Burden of Disease project estimated that exposure to $PM_{2.5}$ in 2005 contributed 3.2 million deaths and 76 million disability-adjusted life years (DALYs) globally, making ambient $PM_{2.5}$ the 9th most important health risk factor investigated (Lim et al., 2012). In the same study, exposure to ambient O_3 in 2005 was estimated to contribute 0.15 million deaths and 2.4 million DALYs globally.

Evidence of the effects of short-term exposure to PM_{10} suggests small relative increases in daily mortality rates, on the order of 0.05 and 0.1 percent increase in daily mortality per 1 µg m^{-3} PM_{10}. There are fewer reports for $PM_{2.5}$ associations. However, the available evidence from meta-analysis and multi city studies around the world are generally consistent, suggesting associations of around 0.1% per 1 µg m^{-3} $PM_{2.5}$ (TF-HTAP 2011). Examining the effects of long-term exposure to PM, Krewski et al. (2009) report an increased relative risk of all-cause mortality of 0.3% (95% CI of 0.1-0.5%) per 1 µg m^{-3} $PM_{2.5}$.

Atkinson et al. (2012) summarised recent studies of risk estimates for all-cause mortality due to short-term exposure to O_3 to compare with their estimates for 5 urban and 5 rural areas of the UK of 0.48% (95% CI: 0.35, 0.60) and 0.58% (95% CI: 0.36, 0.81) per 10 µg m^{-3} increase in O_3, respectively. A systematic review for the UK Department of Health

reported a meta-analytic summary estimate for deaths from all-causes (20 single-city estimates) of 0.22% (0.09%, 0.35%) (Anderson et al. 2007). The comprehensive re-analysis of data from multiple cities in the US, Canada and Europe (Katsouyanni 2009) reported all-year coefficients for all-cause mortality per 10 μg m^{-3} increments in mean O_3 (lags 0 and 1), of 0.32% (95% CI: 0.12%, 0.52%) across 54 US cities, 0.97% (0.67%, 1.3%) across 12 Canadian cities and 0.12% (–0.02%, 0.26%) across 23 European cities. These are in line with previous reviews by Bell (2004, 2005) and Smith et al. (2009) whose estimates of the associations between O_3 and mortality range from 0.3% to 0.6% per 20 μg m^{-3} increments in mean O_3. As discussed in section 2, the evidence for O_3-related mortality due to long-term exposure is increasing (e.g. Krewski et al. 2009; Jerrett et al. 2009). Krewski et al. (2009) give risk estimates for all-cause mortality of 1.02 (95% CI: 1.01-1.03) per 20 μg m^{-3} summertime O_3 whilst Jerrett et al. (2009) found only risk estimates for respiratory mortality in summer of 1.04 (1.013-1.067) per 20 μg m^{-3} (WHO, 2013). Hence much uncertainty remains in the effect of long-term exposure to O_3.

4.2 Current air pollution health impacts attributable to climate change

Long-term historic trends in spatiotemporal surface concentrations of air pollutants have principally been driven by the changes in the anthropogenic emissions of the primary pollutants and of the precursors from which the secondary components of air pollutants are derived. For example, simulations of changes in O_3 since pre-industrial times show that surface O_3 has increased everywhere. Throughout the northern hemisphere >40% of present-day O_3 is due to post-industrialisation activity. No study has sought to estimate additional changes in health impacts from air pollution arising from climate change over the industrialised period. Such attribution is difficult because of the large number of mechanisms by which climate change can influence trends in surface concentrations of air pollutants and the difficulty in attributing changes in relevant factors specifically to anthropogenic climate change vs. decadal or interannual variability.

In terms of future trends, Stott et al. (2004) estimate with > 90% confidence that human influence has at least doubled the risk of a heatwave equivalent to that experienced across Europe in 2003, whilst Jaeger et al. (2008) report that anthropogenic climate change has contributed more than 90% to the probability of the same summer heatwave in the Alps region. As discussed in section 3 it is generally concluded that summer 2003 could represent “normal” summers in the future. In 2003, up to one-third of the excess deaths occurring during the heatwave in the UK were estimated to derive from exposure to high levels of O_3 and PM_{10} during this period (Stedman, 2004, Johnson et al. 2005). In a similar study for the Netherlands, Fischer et al. (2003) found an excess of 1000-1400 deaths during the summer of 2003, with 400-600 of these deaths related to O_3 and PM_{10}. For nine cities in France (where the effects of the 2003 heatwave were greatest), Filleul et al. (2006) estimated the excess risk of death was significant (1.01%; 95% CI, 0.58-1.44) for an O_3 increase of 10 μg m^{-3}, and highlighted the joint risk due to temperature and O_3. However, it is not possible to equate the probabilistic attributions of anthropogenic climate change to the high temperatures during this heatwave with attributions of climate change on pollutant concentrations during this period.

The most comprehensive assessment of the anthropogenic influence on air pollution is from Fang et al. (2013) who sought to attribute the change in global premature mortality associated with change in O_3 and $PM_{2.5}$ between pre-industrial (1860) and 2000 between three factors: emissions of short-lived air pollutants, climate change and increased methane (CH_4) concentrations. They report global mean surface $PM_{2.5}$ and health-relevant O_3 (defined in their study as the maximum 6-month mean of 1-h daily maximum O_3 in a

year) increased by 8 ± 0.16 µg m^{-3} and 30 ± 0.16 ppbv (annual average ±standard deviation of 10-y model simulations), respectively, over this industrial period as a result of combined changes in emissions of air pollutants (EMIS), climate (CLIM) and CH_4 concentrations (TCH4). EMIS, CLIM and TCH4 caused global population-weighted average $PM_{2.5}$ (O_3) to change by +7.5 ± 0.19 µg m^{-3}, +0.4 ± 0.17 µg m^{-3}, and 0.04 ± 0.24 µg m^{-3}, respectively. Total global changes in $PM_{2.5}$ were associated with 1.5 (95% CI, 1.2-1.8) million cardiopulmonary mortalities and 95 (95% CI, 44-144) thousand lung cancer mortalities annually and changes in O_3 were associated with 375 (95% CI, 129-592) thousand respiratory mortalities annually. The results show that most of the increase in air pollution mortality has been driven by changes in emissions of short-lived air pollutants and their precursors (95% and 85% of mortalities from $PM_{2.5}$ and O_3 respectively). Changing climate contributed up to 5% of the premature mortality associated with air pollution globally, and by more in some regions. Silva et al. (2013) also estimate the global burden of anthropogenic outdoor air pollution of O_3 and $PM_{2.5}$ on present-day premature human mortality, and the component of that burden attributable to past climate change using modelled concentrations from an ensemble of chemistry–climate models. They estimate that, at present, 470, 000 (95% confidence interval, 140,000 to 900, 000) premature respiratory deaths are associated globally and annually with anthropogenic O_3, and 2.1 (1.3 to 3.0) million deaths with anthropogenic $PM_{2.5}$-related cardiopulmonary diseases (93%) and lung cancer (7%).

Overall, however, as demonstrated by Fang et al. (2013), and by analogy with model simulations of the comparative effects into the future of potential emissions changes and climate change (see section 5.3), it is anticipated that the changes in health impacts of air pollution over the last 40 years that are in principle, attributable to anthropogenic climate change is much smaller than the changes in health impacts attributable to changes in emissions.

5. Air pollution related health impacts in the future

In this section the results from studies of the effects of climate and emissions changes on future air quality over Europe are reviewed both in terms of the independent effects and the combined effects of emission scenarios and climate change. Absolute changes in future O_3 or PM levels are discussed only from multi-model estimates or from the most recent literature. Health impacts where assessed have been included in the relevant section.

5.1. Future Climate Scenarios

In the third (TAR) and fourth (AR4) Intergovernmental Panel on Climate Change (IPCC) assessment reports (IPCC, 2001; 2007), the Special Report on Emissions Scenarios (SRES; Nakicenovic et al., 2000) formed the basis of future climate change projections. These covered a wide range of demographic, economic and technological driving factors, resulting in a range of greenhouse gas (GHG) emissions. Based on the AR4 multi-model ensemble of climate projections, the range of global mean warming for the years 2090-2099 relative to 1980-1999 is 1.4-6.3°C for these SRES scenarios (IPCC, 2007).

A new set of emission scenarios, the "Representative Concentration Pathways" (RCPs) were developed in support of IPCC's fifth assessment report (AR5- IPCC, 2013). There are 4 RCPs covering a range of net radiative forcing projections at the top of atmosphere (TOA) out to 2100: RCP2.6 (vanVuuren et al., 2011), RCP4.5 (Thomson et al., 2011), RCP6.0 (Masui et al., 2011), and RCP8.5 (Riahi et al., 2011). In contrast to SRES, some of the RCPs consider climate mitigation and stabilization. A time series of historical and projected temperature changes for the RCPs from AR5 are shown in Figure 4. By the end of the century, the increase of global mean surface temperature above 1986-2005 levels is projected to be: 0.3-1.7˚C for RCP2.6 and 2.6-4.8˚C for RCP8.5.

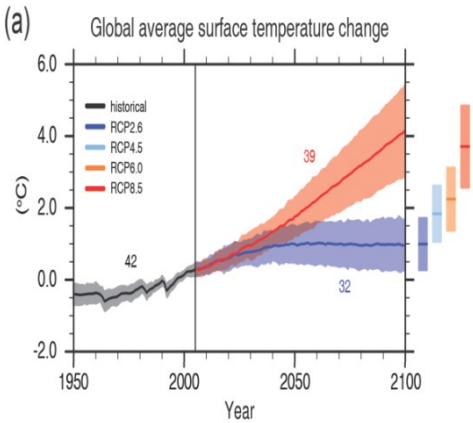

Figure 4: CMIP5 time series from 1950 to 2100 of global annual mean surface temperature relative to the 1986-2005 time period. The projections out to 2100 are based on RCPs 2.6 and 8.5. The shading represents one standard deviation and the number of models is given in the same colour. The projected global annual mean temperature change for 2081-2100 relative to 1986-2005 and the associated standard deviations for the 4 RCPs are shown as coloured vertical bars to the right of the figure.

Jacob and Winner (2009), Isaken et al. (2009) and Fiore et al. (2012) reviewed the impact of climate change, in the absence of emission changes, on air quality. In particular, they noted that O_3 is strongly correlated with temperature (Cox and Chu, 1995) due to associations of temperature with enhanced photochemical reaction rates, in particular enhanced PAN decomposition (Sillman and Samson, 1995; Racherla and Adams 2008; Doherty al. 2013), leading to local NO_2 and O_3 increases in the source region, as well as stagnation events, and elevated natural emissions from biogenic and wild fire sources. A number of these factors were established as the cause of high levels of O_3 during the 2003 heatwave (e.g. Vautard et al. 2005; Solberg et al., 2008; Vieno et al. 2010; see section 3). Doherty et al. (2013) highlight the crucial role of temperature sensitivity to isoprene finding it to be the dominant mechanism for increasing O_3 levels in polluted regions. Racherla and Adams (2008) also find enhanced isoprene emission in future climate to be the dominant cause of increased summer O_3 chemical production in the eastern USA. However, the extent to which CO_2 inhibition of isoprene emissions in a future higher CO_2 climate may offset temperature-driven emission increases may be substantial (Rosenstiel et al., 2003; Arneth et al., 2007; Heald et al., 2009; Young et al., 2009; Squire et al. 2014). In addition, Ito et al. (2009) and Fiore et al. (2012) highlight that the sign of the O_3 response to temperature and climate change depends on the assumption of the amount of recycling of NO_x from isoprene nitrates.
Another important global O_3 precursor, CH_4, has a large natural emissions source from wetlands; these emissions are likely to increase under climate change along with CH_4 emissions from other sources as a result of climate feedbacks (e.g. permafrost thaw) (O'Connor et al. 2010). Forkel and Knoche (2006) also suggest a minor role for temperature, through higher soil NO_x emissions, to increase summertime mean daily maximum surface O_3 concentrations in agricultural areas in Europe.

The overall effect of temperature-driven processes on PM is even more complex to disentangle than for O_3 due to opposing influences on various PM components (Fiore et al. 2012). Large feedbacks are possible due to elevated emissions from biogenic and wild fire sources, yielding carbonaceous particles, mineral dust and SOA. The sulphate and nitrate components respond differently to elevated temperature (and humidity) as discussed in section 3. SO_2 oxidation to sulphate aerosol is enhanced, increasing sulphate aerosol (e.g. Hedegaard et al. 2008), whilst increased partitioning to the gas phase reduces nitrate and

some organic aerosol species (e.g., Racherla and Adams, 2006; Pye et al. 2008; Jacob and Winner 2009; Fiore et al. 2012). In particular, several studies over the USA suggest large decreases of nitrate PM with increasing temperature, and this is the dominant effect on PM concentrations in regions where nitrate is a relatively large component (Dawson et al. 2007; Jacob and Winner, 2009), as is often the case in urban UK regions.

Increased atmospheric water vapour with elevated temperature will increase O_3 destruction in low-NO_x environments (e.g. Johnson et al., 1999), causing a reduction in global background concentrations of surface O_3. In addition, studies of recent heatwaves discussed above (Vautard et al. 2005; Solberg et al. 2008; Vieno et al. 2010, Emberson et al. 2013) also highlight that under drier future climate conditions, reduced soil moisture leads to suppressed O_3 dry deposition and higher surface O_3. For PM, increased water vapour also leads to higher OH levels contributing to enhanced SO_2 oxidation and higher sulphate concentrations outlined above. Nitrate levels also increase with higher humidities (Dawson et al. 2007). PM concentrations decrease in areas simulated to have increased precipitation frequency as wet deposition provides the main sink (e.g. E. USA; Penrod et al. 2014). Under climate change it is generally thought that wetter regions of the world will get wetter and drier regions drier (Held and Soden, 2006; Liu and Allen 2013). Changes in cloud amount have also been shown to influence surface O_3. Several authors report increased summertime surface O_3 concentrations over Europe and the NE USA due to reduced cloud amounts that lead to enhanced photolysis rates (Meleux et al, 2007; Kunkel et al., 2008; Katragkou et al., 2011), particularly that of NO_2 which favours O_3 formation (Forkel and Knoche 2006).

Climate change also has the potential to alter transport pathways (Mickley et al. 2004) and mixing depth (Penrod et al, 2014). Convection, for example, is expected to be deeper, although less frequent under climate change, and chemistry transport model simulations typically suggest that lightning NO_x emissions will increase (e.g. Hauglustaine et al., 2005, Young et al. 2013; Banjeree et al. 2014). Likewise, climate change is predicted to increase stratosphere-troposphere exchange as a result of an enhanced Brewer Dobson circulation (Butchart and Scaife, 2001) which increases the stratospheric contribution to surface O_3 (Collins et al. 2003; Lacressonnière et al. 2015). Recent simulations by Glotfelty et al. (2013) have shown that climate change may enhance intercontinental transport of air pollution from East Asia increasing global average O_3 and $PM_{2.5}$ slightly. More recently, Doherty et al. (2013) suggested that shifts in transport patterns associated with inter-hemispheric transport are unlikely to have a major role in influencing the annual-mean O_3 response due to climate change. However, they may well be important when considering changes in extremities of O_3 concentrations e.g. the 95% percentile or in daily 8-hour maximum surface O_3. Wu et al. (2008b) also found O_3 pollution episodes were far more affected by climate change (e.g., frontal passages) than mean values. In relation to mixing depth, a number of climate model projections for the 21st century provide inconsistent results on the impact of climate change on mixing depth, with increases and decreases in different regions (e.g. Mickley et al., 2004; Murazaki and Hess, 2006; Wu et al., 2008a, Penrod et al. 2014). In the recent IPCC's AR5 report, it was concluded that there is limited evidence and agreement on the impact of climate change on transport pathways (IPCC, 2013). Although the current generation of chemistry-climate models include these processes, a comprehensive assessment of their relative impacts is still lacking (Kirkman et al. 2013).

Figure 5 depicts the global and regional surface O_3 response to climate change alone between 2000 and 2030, where the range reflect multi-model differences in spatial averages (solid green lines) and spatial variability within a single model (dashed green

lines). The decrease in global mean O_3 is driven by higher water vapour and temperatures (high confidence). The higher temperatures can lead to local O_3 increases during the peak pollution season (2-6 ppbv) for Central Europe (green dashed line based on Forkel and Knoche, 2006). Overall, there is medium confidence based on observations and modelling that a warmer climate will increase O_3 in polluted regions (Kirtman et al., 2013), the so-called "climate penalty effect" (Wu et al. 2008b, Bloomer et al. 2009; Rasmussen et al. 2013). $PM_{2.5}$ concentrations are expected to decrease in regions where precipitation increases (medium confidence). However, there is a lack of consensus on other climate-driven factors leading to low confidence in the overall impact of climate change on $PM_{2.5}$ distributions (Kirtman et al., 2013).

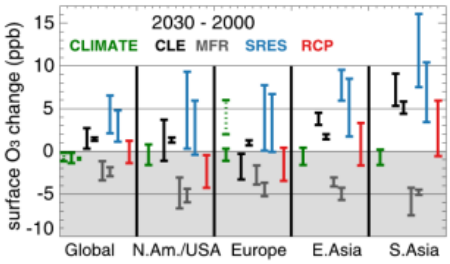

Figure 5: From chapter 11, IPCC WG1 Fifth Assessment Report, figure 11.22, adapted from Fiore et al. (2012). Changes in surface O_3 (ppb) between year 2000 and 2030 driven by climate alone (CLIMATE; green) or emissions alone following CLE (black), MRF (gray), SRES (blue) and RCP (red) emission scenarios. Bars represents multi-model standard deviation. For further details see Kirtman et al. (2013).

Studies of climate change impacts on air quality in 2100 suggest the same effects of background O_3 and O_3 in polluted regions as described above, but the impact is typically larger with the greater climate signal. In a multi-model study, Doherty et al. (2013) suggest annual-mean O_3 increases of up to 6 ppbv in polluted regions reaching up to 14 ppbv in the season of maximum O_3. In the recent Atmospheric Chemistry and Climate Model Intercomparison Project (ACCMIP), Young et al. (2013) suggest annual-mean O_3 increases of >10 ppbv over Europe (see their Fig 9) under RCP 8.5 in 2100 compared to 2000. In a regional European multi-modelling study to 2100 using the SRES A1B scenario Langner et al. (2012a) report that, in southern Europe, climate change leads to increased summer mean O_3 of 0-3 ppbv and increased summer daily maximum O_3 of 3-6 ppbv. In northern Europe they found reductions of 0-3 ppbv for both mean and daily maximum O_3 in summer. Langner et al. (2012b) suggest that climate change has greater impact on episodic O_3 (they examine the 95th percentile of hourly O_3) than on longer-term (mean and daily maximum O_3) summer averages. Collette et al. (2013) also found similar geographical patterns of projected impact of climate on O_3 with an increase over southern continental Europe and a decrease over northern Europe and the British Isles.

There have been only a few studies to date examining the effect of climate change alone on human health burdens. Fang et al. (2013) quantified the effect of climate change induced changes in PM2.5 and O3 air quality over the 21st century under the moderate SRESA1B scenario on global premature mortalities. PM2.5 concentrations increases due to elevated sulphate concentrations and reduced precipitation over the major emission source regions, led to an increase in global annual premature mortality associated with chronic exposure to PM2.5 of approximately 100,000 deaths (95 % confidence interval, CI, of 66–130,000) with corresponding years of life (YLL) lost increasing by nearly 900 thousand (95 % CI, 576–1,128 thousand) years. Higher O3 in polluted regions also increased annual premature mortality due to respiratory disease from chronic O3 exposure by +6,300 deaths (95 % CI, 1,600–10,400). On average, across 50 U.S. cities, higher O3 under the SRES A2 high climate scenario increased total daily mortalities by 0.11–0.27 % from 2000s to 2050s (Bell et al. 2007). For the same period Tagaris et al. (2009) estimated that climate change under the SRES A1B scenario increased combined PM2.5 and O3 related annual U.S premature mortalities by 4,300 deaths. Over Europe, Orru et al. (2013), estimated annual premature mortalities to increase over most of Europe but decrease over

the northernmost Nordic and Baltic countries, with the largest change being a 34% increase over Belgium under the SRES A2 climate scenario, due to regional reductions in cloud cover and soil moisture. In a sensitivity study over the UK, Heal et al. (2013) found that a 5°C increase of year-round temperatures increased the total UK health burden by an additional 500 premature deaths (4%), assuming no change in population and no threshold for O3 effects.

5.2. Future Anthropogenic Emission Scenarios

The SRES and RCP emission scenarios for greenhouse gases used for the climate projections discussed in section 5.1 additionally provide emission scenarios for short-lived precursor species of surface O_3 and secondary PM. The SRES scenarios span a wide range of future change in air pollutant emissions. For example, projected emissions of nitrogen oxides (NO_x) from anthropogenic sources for the year 2100 are in the range of 18.7-109.6 Tg(N) y^{-1} (*cf.* Year-2000 anthropogenic emissions of 37.2 Tg(N) y^{-1} from Lamarque et al., 2010). The corresponding range for anthropogenic carbon monoxide (CO) emissions is 363-2570 Tg(CO) y^{-1} (*c.f.* Year-2000 emissions of 608.7 Tg(CO) y^{-1} from Lamarque et al., 2010).

Wild et al. (2012) estimated regional average changes in surface O_3 due to changes in anthropogenic precursor emissions under the SRES scenarios based on source receptor calculations with 14 different global models. Changes in annual mean O_3 between 2000 and 2050 averaged over Europe were: –1.2 ppbv (B1), 6.2 ppbv (B2), 4.6 ppbv (A1B) and reaching 7.7 ppbv under the SRES A2 scenario.

Other studies have made use of alternative emission scenarios derived from the SRES scenarios, for example, current legislation (CLE) and maximum feasible reductions (MFR) derived by IIASA (the International Institute of Applied Systems Analysis) (Dentener et al. 2005). CLE is based on the IPCC SRES B2 socioeconomic scenario, plus adherence to emissions reduction air quality legislation prevailing in the year 2000. MFR is also based on SRES B2, but including maximum feasible reductions in emissions achievable through implementation of all abatement measures available in 2000. These scenarios only considered projected changes in O_3 precursor emissions out to 2030, are lower than the SRES scenarios, and tend to cover the RCP range. Dentener et al. (2006) performed a multi-model comparison of 26 global CTMs to examine the effects of the SRES A2 and CLE and MFR scenarios on O_3 air quality in 2030. Increases in ensemble-mean surface O_3 over the UK of 2-4 ppbv and 2-6 ppbv occurred under the CLE and SRES A2 scenarios, whilst a change of –1 to +1 ppbv was reported under the MFR scenario. However, the coarse resolution of these global models combined with the ensemble averaging may not capture effects of O_3 removal through reaction with NO in NO_x-saturated areas (known as titration). Indeed, for these same emissions scenarios but using a higher resolution regional model, Heal et al. (2013) found markedly different results for annual-mean changes in O_3 over the same time period: for the B2+CLE scenario, O_3 increased by 1.5-3 ppbv everywhere over the UK; for the SRES A2 scenario, O_3 decreased over most of England, reaching –2 ppbv in urban areas and –3 ppbv in the London area, and increased by 0-3 ppbv everywhere else; and, for the B2+MFR scenario, the reverse of the pattern under SRES A2 (increases of 0-3 ppbv over most of England, plus S. Wales, Edinburgh-Glasgow and Belfast, and decreases up to –1.5 ppbv elsewhere). These changes in UK surface O_3 reflect differences in the amount of background O_3 imported to the domain, in conjunction with differences due to changes in UK O_3 precursor emissions that influence the extent of O_3 titration in high NO_x (i.e. urban) regions. Markakis et al. (2014) also find that NO titration is strongly influenced by model

resolution using a high resolution model for Paris. Both studies highlight the VOC-limited chemistry that occurs at the higher resolution in urban areas; whilst at regional-scale NO_x-limited conditions may prevail. The effect of resolution on the chemical environment is likely more important in winter than in summer (Stock et al. 2014).

Based on the above, Heal et al. (2013) also estimated O_3-related health burdens for the UK for three anthropogenic emissions scenarios for 2030 on a 2003 baseline. Including UK population changes in 2030, both the CLE and MFR scenarios yielded greater O_3-attributable health burdens than the SRES A2 emission scenario with increases of: 28%, 22%, and 16%, respectively, above 2003 baseline deaths brought forward (11,500) and respiratory hospital admissions (30,700), using O_3 exposure over the full year and no threshold for health effects. The health burdens were greatest under the CLE scenario because O_3 concentrations increase as a result of both increases in background O_3 concentration and decreases in UK NO_x emissions. As for the perturbed temperature scenario in this study (section 5.1), if a 35 ppbv threshold for O_3 effects was assumed, health burdens were roughly an order of magnitude lower. Likhvar et al. (2015) also found O_3-related mortality to increase under CLE and MFR scenarios extended to 2050 for Paris. Their assessment of the combined effect of O_3 and $PM_{2.5}$ mortality showed that MFR consistently led to greater health benefits compared to CLE across multiple geographical scales (Europe and Ile-de-France).

The 4 RCPs developed for IPCC AR5 (IPCC, 2013) also considered emissions relevant to air quality, as well as climate mitigation and stabilisation. They all assumed substantial reductions in aerosol and O_3 precursor emissions. The only exceptions to this are CH_4 in RCP8.5, which doubles relative to year-2000 concentrations and ammonia (NH_3) which increases in all pathways. However the RCP scenarios were primarily developed to encompass a range of long-term climate change outcomes and do not specifically consider uncertainties in air pollution development in the shorter term (Collette et al. 2012). In particular, NO_x emissions are similar in all RCP scenarios except RCP 8.5 (Cionni et al. 2011). Wild et al. (2012) also calculated estimates of changes in annual mean O_3 under the RCP scenarios between 2000 and 2050 averaged over Europe. These changes were negative (−2 to −4.7 ppbv) for all RCP scenarios except for RCP 8.5 (+0.3 ppbv) which has the largest increase in CH_4. An average difference of ~ 5ppbv between the RCP 2.6 and RCP 8.5 scenarios was reported, of which 75% was attributed to differences in CH_4 abundance. The changes were substantially different than estimated with the SRES A1B, A2 and B2 scenarios, reflecting the assumptions of more stringent O_3 precursor emission controls under the RCP scenarios.

Recently new “Global Energy Assessment (GEA) air quality” scenarios have been developed to accompany the RCP scenarios and used in European modelling studies (Rao et al., 2012; Riahi et al., 2012). Under these two GEA scenarios NO_x emissions in 2050 are reduced to 30-50 % of their current (2005) levels. Using these emission scenarios with 6 regional and global CTMs, Collete et al. (2013) reported increases in ensemble-median annual mean O_3 in NO_x-saturated areas of ~5-10 ppbv which includes most UK cites (see their fig 5) accompanied by large decreases in southern Europe. This highlights the importance of considering VOC as well as NO_x controls for O_3 policy legislation.

Finally, for the Task Force on Hemispheric transport of air Pollution (HTAP), Anenberg et al. (2009) estimated the effect of regional-wide 20% emission reduction of O_3 precursors on O_3 and $PM_{2.5}$ related mortality. They suggest that EU emission reductions lead to more avoided mortalities outside the EU than within the region due to the higher populations

downwind e.g. over Africa. In another study they estimate that halving global anthropogenic black carbon emissions avoids 157,000 annual premature deaths globally, mostly near the source region (Anenberg et al. 2011). Much of the literature considers the combined effects of these emissions scenarios in terms of precursor emission and climate change on O_3 and PM, and these are described in the next section.

5.3 Impacts of Future Climate and Emissions on Air Quality

A number of recent studies have sought to evaluate future changes in air pollution, for surface O_3 in particular. Whilst some of these studies have split the impacts on air quality between climate and emissions, these attributions are difficult for several reasons: global-to-regional downscaling of meteorology is model dependent, the brief simulations preclude separation of climate change from climate variability, and the lack of systematically explored standard scenarios for emissions (Kirtman et al. 2013). The impacts of climate change on air pollution are also likely to depend on the exact future trends in magnitudes of anthropogenic emissions.

Langner et al. (2012a) studied air quality in 2100 under the SRES A1B climate scenario together with O_3 precursor emission changes from the RCP 4.5 scenario. The impact of reduced emissions in Europe dominated over climate change impacts and led to reductions of April-September daily max O_3 both in northern and southern Europe in 2100. They found in southern Europe projected emissions reductions more than offset the climate change related increase in O_3 in summer. In northern Europe, both emission reductions and climate change decreased O_3. Reductions in April-September daily maximum O_3 of −9.2 and −12.8 ppbv century^{-1} were reported for north and south Europe respectively. Coleman et al. (2013) reported similar finding for the RCP 6.0 scenario with the most significant changes after 2050 due to the pattern of changing emissions. They also noted the importance of changes in meteorology over the N. Atlantic region over the longer time horizon of 2100. Under the RCP 8.5 scenario Lacressonnière et al. (2014) reported that the higher CH_4 emissions results in increased surface O_3 over NW Europe but decreased surface O_3 over southern parts of Europe in both the 2030s and 2050s.

Collette et al. (2013) examined the combined effects of the RCP 2.6 and RCP 8.5 scenarios alongside compatible GEA scenarios on air quality in 2050 for both O_3 and $PM_{2.5}$. For both pollutants, they also showed that the main factor driving future air quality projections for Europe was again air pollutant emissions, rather than climate change or intercontinental transport of pollution. The geographical patterns of projected impact of climate on ozone were similar to that found by Langner et al. (2012), and a clear O_3 climate penalty was found across most of Europe. The geographical patterns of the impact of climate on PM appeared much less robust (Collette et al. 2013). The resulting population-weighted SOMO35 O_3 health metric (defined as the annual sum of daily maximum O_3 over 35 ppbv based on 8 h running means) change varied between +7% (reference scenario) to –80% (mitigation scenario). Annual mean $PM_{2.5}$ was reduced in both scenarios which resulted in a 62-78% reduction in exposure-weighted $PM_{2.5}$. However, the authors point out the sensitivity to the emission scenarios used and to precipitation projections. Kim et al. (2015) also noted the potential of increased natural NO_x emissions from lightning and soils to offsets anthropogenic NO_x emission reductions.

The combined effects of emission and climate changes under the RCPs spatially averaged over Europe based on IPCC AR5 (Kirtman et al. 2013) are shown in Figure 6. By 2100, European multi-model annual-mean surface O_3 changes between –15 to +2 ppbv relative to 2000 and $PM_{2.5}$ changes between –4 to –6 μg m^{-3}. The increases under RCP 8.5 reflect

primarily the increase in CH_4 emissions discussed above (Fiore et al. 2012, Wild et al. 2012), whilst the decreases in the other three scenarios reflect the role of emission reductions of O_3 precursors. PM changes depend on oxidant levels but generally follow SO_2 emissions and primary organic carbon emissions changes (Fiore et al. 2013, Kirtman et al. 2013). Competition between sulphate and nitrate for ammonium means that reducing SO_2 whilst increasing NH_3 emissions as in these RCPs would lead to nitrate aerosol levels equal or higher than those of sulphate in the near term (Kirtman et al., 2013). All RCP scenarios suggest similar decreases in European-average $PM_{2.5}$. Overall, the emissions changes strongly drive changes in O_3 and $PM_{2.5}$ in terms of annual-mean metrics. These are either augmented or reduced by the multiple effects of climate on atmospheric composition. However, for other metrics relating to peak exposure levels the effects of changes in climate may be relatively more important.

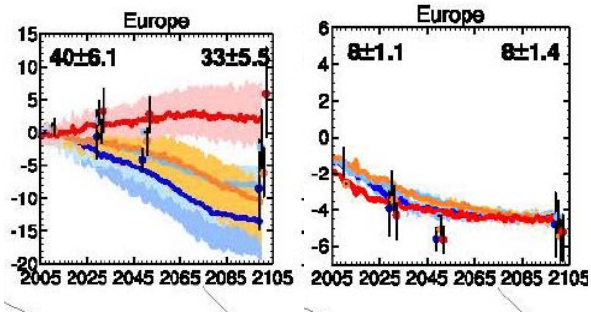

Figure 6: Projected changes in annual-mean (left) O_3 (ppbv) and (right) $PM_{2.5}$ (μg m^{-3}) from 2000 to 2100 following the RCP scenarios (8.5 red, 6.0 orange, light blue 4.5, 2.6 dark blue) averaged over Europe (land). Coloured lines show the average and shading denotes the full range of 4 chemistry-climate models and coloured dots and bars represent the average and full range of ~15 ACCMIP models. Taken from Fiore et al. (2012) as used in Kirtman et al., (2013).

In terms of O_3 health burdens, Heal et al. (2013) also reported greater O_3 penalties from emissions changes compared to temperature change. Using the RCP scenarios and results from ACCMIP, Silva et al. (2015 in prep) suggest that global $PM_{2.5}$ mortality generally decreases in the future, but O_3 mortality increases in some scenarios/periods.

5.4. Future Changes in Extremes and Air Quality

Air pollution events/episodes are typically associated with stagnation events (which may or may not be associated with blocking high pressure systems) and sometimes with heatwaves (Fiore et al. 2012). As discussed above climate change has increased the near term risk of heatwaves which can often be associated with O_3 exceedance days (Patz et al 2014) through a number of processes as discussed in section 3.

The impact of climate change on the frequency of blocking episodes diagnosed from multiple climate model simulations for IPCC AR5 under RCP8.5 show a consistent decrease in winter and summer blocking over Europe in the 21st century (Masato et al. 2013). However, in summer the accompanying poleward shift of the storm track into the region of frequent high-latitude blocking, may lead to a greater number of storms obstructed by blocking high pressure systems in the future (Masato et al. 2013). It is also noted that these results depend on the accurate simulation of present-day blocking (Massato et al. 2013). Changes in large-scale blocking may affect local stagnation and heatwave episodes. However the relationship between blocking and stagnation remain unclear (Webber et al. 2015 in prep). Indeed, Horton et al. (2014) report changes in stagnation under RCP8.5 using an air stagnation index based on bias correct dry day and surface and upper level wind speeds. Annual-mean and springtime stagnation occurrence increases over Mediterranean Europe by the late 21st century. Other seasonal changes were not reported. The authors noted that biases in modelled surface wind speeds were high. Overall Kirkman et al. (2013) assign low confidence in projecting changes in meteorological blocking associated with extreme air pollution episodes.

There is high agreement from modelling studies projecting increases in the frequency and duration of extreme O_3 pollution events but there is large variability in predicted spatial pattern and incidence of these events (Forkel and Knoche 2006; Fiore et al. 2012 and references therein, Kirtman et al., 2013). However, the collective evidence suggests that increasing temperatures during air stagnation episodes in polluted environments will increase peak pollution (medium confidence) (Fiore et al. 2012; Kirtman et al. 2013).

5.5 Key non-climate/emissions determinants

The effects of higher air pollution levels and air pollution episodes will undoubtedly be modified by behaviour that affects exposure such as time indoors and exercise that affects inhalation. These are discussed further in section 6.

The extent of adverse health impact of exposure to surface O_3 depends markedly on whether a threshold concentration of O_3 below which no impact is assumed and, if so, the magnitude of this threshold (Royal Society 2008, Atkinson et al. 2012, Heal et al. 2013). As discussed in section 5.2, Heal et al. (2013) found that assuming O_3 exposure over the full year to be relevant, the attributable health impact for a 35 ppbv threshold is approximately only 10% of that attributable if no threshold is assumed. The health impact assuming a 50 ppbv threshold was found to be approximately a further factor 5 lower than that for the 35 ppbv threshold.

There are also uncertainties in the appropriate magnitude of exposure-response coefficient to use, and the part of the year for which O_3 exposure is relevant for health impact (WHO 2013). Uncertainties in risk estimates were found to be as large an uncertainty as simulated O_3 concentrations for assessing the health effects of O_3 precursor emissions (Anenberg et al. 2009). Anenberg et al. (2011) also suggest that the choice of concentration-response factor and health effect thresholds affects estimated global avoided deaths due to exposure to black carbon by as much as 56%, but does not strongly affect the regional distribution.

Issues concerning potential modification by temperature remain pertinent especially for future assessment (Filleul et al. 2006; Ren et al. 2008; Pattenden et al., 2010; Atkinson et al., 2012) and to seasonally-varying relationships with PM. It has also been assumed that regional daily baseline mortality and morbidity rates remain constant in the future (Heal et al. 2013). It is not possible to predict changes in concentration–response coefficients and threshold effects of any autonomous or planned adaptation to future O_3 levels or to future climate change (Knowlton et al., 2004).

6. Potential for impacts to be avoided by adaptation measures

Research performed under the auspices of The Task Force on Hemispheric Transport of Air Pollution (TF-HTAP) has focused on quantifying the need for international co-operation on controls of air pollutants (TF-HTAP, 2011). In their recommendations they suggest that "Our current understanding of the magnitude of intercontinental flows of air pollution is sufficient to conclude (a) that such flows have a significant impact on environmental quality throughout the Northern Hemisphere and (b) that coordinated international actions to mitigate these flows would yield significant environmental and public health benefits." In addition, air quality limit and target values for air pollutants are devised for each species separately. Dominici et al., (2010) recommend that legislation that considers multiple

pollutants can help identify the most effective control measures to reduce the overall adverse human health burdens.

Outside of changes to legislation, a number of potential adaptations at a hemispheric to local scale can be highlighted. There is some evidence for physiological adaptation to O_3 from a study that reported diminished mortality effects later in the summer, reaching the null effect by September (Zanobetti and Schwartz, 2008). In a warmer future, more windows may be open, or individuals may spend greater time outdoors than indoors. On the other hand, there may be increased use of air-conditioning leading to decreased exposure to ambient O_3 and PM. Further discussion of responses to increased temperatures and as mediated via the built environment is beyond the scope of this report.

With regard to accurate forecasting in the UK, for heatwaves a joint Met Office/Department of Health project called the Heat-Health Watch gives advanced warning of hot weather in the UK (http://www.metoffice.gov.uk/weather/uk/heathealth/). It has four levels of response based upon threshold maximum daytime (two days) and minimum night-time temperatures (intervening night) which vary regionally, but an average threshold temperature is 30 °C by day and 15 °C overnight. The UK government Department for Environment, Food and Rural Affairs (Defra) also issues a daily air pollution index forecasts for the UK. The index is numbered 1-10 and divided into four bands, low (1) to very high (10), to provide detail about air pollution levels (http://uk-air.defra.gov.uk/forecasting/). The MACC project also issue daily air pollution forecasts across Europe using a suite of seven regional models to provide an ensemble forecast for real-time and up to 3-day forecasts (http://www.gmes-atmosphere.eu/services/raq/raq_nrt/). These forecasts could be used as the basis for triggering the adoption of adaptation measures among the ‘at risk’ population.

7. Conclusions and Evidence Gaps

A number of recent studies have sought to evaluate changes in future air pollution human health impacts, for surface O_3 in particular. However, considerable uncertainty exists due to the multitude of mechanisms by which changes in anthropogenic and natural emissions and in meteorology/climate can influence O_3 and PM concentrations.

Globally, warming decreases background surface O_3, but higher CH_4 levels (RCP8.5) increase surface O_3 counteracting this decrease. Regionally, several studies suggest fairly robust climate change signals over Europe that lead to increase in summer-mean O_3 in southern Europe and little change in northern Europe. The effects for higher concentration metrics such as daily maximum O_3 are typically larger since higher surface temperatures in polluted regions enhance local chemistry and emissions, increasing peak levels of O_3 and $PM_{2.5}$ but confidence in this is not high. Multi-decadal simulations with fully coupled chemistry-climate models would lead to improved understanding of the interactions between climate and air quality by being able to isolate an anthropogenic climate signal on air quality relative to that from climate variability, which has not been possible in most published studies to date (Fiore et al. 2012). For PM, studies related to climate change are still limited such that the spatial patterns of change over Europe and magnitudes remain uncertain, partly due to model differences in spatial patterns of precipitation frequency. Climate change may alter natural aerosol sources and wet removal, but no confidence level is attached to the overall impact. Further studies with multiple models that examine the effects of climate change on PM using consistent emissions and climate would be most useful, as well as further assessments of the potentially opposing influences of

climate change on different PM components. Collette et al. (2013) also highlight the need to further understand the influence of background levels of O_3 on which emissions controls and climate scenarios are implemented, as these are important for determining the overall pollutants response.

There is consensus in the literature that near-term air quality will be dominated by emission changes rather than changes in climate and/or intercontinental transport. However, for peak concentrations the effect of the latter two processes may be much more relevant. The latest RCP scenarios show the strong influence of future CH_4 levels on air quality. Controlling CH_4 and possibly black carbon are viewed as a win-win for mitigating air quality as well as climate change (Shindell et al. 2012; UNEP 2013; Baker et al. 2015). However, due to the complexities of the processes linking emissions and air quality, it has not been possible to use the present-day period to identify the current air pollution related burden attributable to observed climate change.

Extreme air pollution episodes are associated with changing weather patterns, such as heat waves and stagnation episodes. The substantial body of literature investigating the 2003 European heatwave is useful since climate change has increased the near-term risk of heat waves. These studies have identified various positive feedback mechanisms associated with climate change that potentially have a substantial effect on O_3 and PM air quality: longer residence times, suppressed O_3 deposition, modified isoprene emission and increased forest fires. However air pollution episodes are more generally related to stagnation events which may in turn be related to larger-scale blocking. A systematic evaluation of all the factors driving extreme pollution episodes is still lacking (Kirtman et al., 2013).

Key uncertainties in mechanistic understanding limit our confidence of future projections of air quality and air quality episodes. A major uncertainty regarding isoprene feedbacks is the opposing influences of rising temperature and CO_2, including inhibition, fertilisation and water use effects on leaf area. Further knowledge of leaf processes, in particular their effect on emission and deposition under future climate conditions, is crucial for constraining the magnitude of the O_3 change as well as the PM response (through SOA) to climate change in polluted regions. The impact of climate change on transport pathways and on climate variability is highly uncertain and model dependent. Projected changes in the frequency of regional air stagnation events also remain difficult to assess. Hence improved understanding and prediction of storm track shifts as well as blocking events and their relationship to stagnation is crucial for improved predictability of air pollution episodes. Projections suggest increases in extreme O_3 pollution events over US and Europe, but there is a lack of agreement at the regional level.

For quantification of the effects of air quality changes on health, major uncertainties for O_3 are the impact of long-term exposure and whether there is a threshold concentration (for both short and long-term exposures) below whether there is no effect. Spatial and temporal heterogeneity in risk estimates for O_3 and PM are also an issue for regional-scale modelling of health impacts. Finally, there is uncertainty regarding future potential adaptation effects and the assumption of unchanging risk estimates into the 21st century.

References

AQEG (2007) Air Quality and Climate Change: a UK Perspective. Third report of the Air Quality Expert Group, UK Department for Environment, Food and Rural Affairs, PB12489, London. http://www.defra.gov.uk/environment/airquality/publications/airqual-climatechange/index.htm.

Anderson HR, Atkinson RW, Peacock JL, Marston L, Konstantinou K. 2004. Meta-Analysis of Time-Series Studies and Panel Studies of Particulate Matter (PM) and Ozone (O_3). Report of a WHO Task Group. Copenhagen:World Health Organization. Available: www.euro.who.int/document/e82792.pdf.

Anenberg, S. C., et al.: Intercontinental impacts of ozone air pollution on human mortality, *Environ. Sci. Technol*., 43, 6482–6487, 2009

Anenberg, S. C., Talgo, K., Arunachalam, S., Dolwick, P., Jang, C., and West, J. J.: Impacts of global, regional, and sectoral black carbon emission reductions on surface air quality and human mortality, *Atmos. Chem. Phys*., 11, 7253-7267, doi:10.5194/acp-11-7253-2011, 2011.

Atkinson, R.W., Yu, D., Armstrong, B. G., Pattenden, S., Wilkinson, P., Doherty, R.M., Heal, M. R., Anderson, H.R., Concentration Response Function for Ozone and Daily Mortality: Results from Five Urban and Five Rural U.K. *Populations, Environ Health Perspect*. 2012 October; 120(10): 14111417

Baker, L. H., Collins, W. J., Olivié, D. J. L., Cherian, R., Hodnebrog, Ø., Myhre, G., Quaas, J., and Samset, B. H.: Climate responses to anthropogenic emissions of short-lived climate pollutants, *Atmos. Chem. Phys. Discuss*., 15, 3823-3862, doi:10.5194/acpd-15-3823-2015, 2015.

Banerjee, A., Archibald, A.T., Maycock, A. C., Telford, P., Abraham, N. L., Yang, X., Braesicke, P., and Pyle, J. A. (2014), Lightning NO_x, a key chemistry–climate interaction: impacts of future climate change and consequences for tropospheric oxidising capacity, *Atmos. Chem. Phys*., 14, 9871-9881, doi:10.5194/acp-14-9871-2014, 2014.

Bell ML, Dominici F, Samet JM. 2005. A meta-analysis of time-series studies of ozone and mortality with comparison to the National Morbidity, Mortality, and Air Pollution Study. *Epidemiology* 16:436–445.

Bell ML, McDermott A, Zeger SL, Samet JM, Dominici F. 2004. Ozone and short-term mortality in 95 U.S. urban communities, 1987–2000. *JAMA* 292:2372–2378.

Bell ML, Goldberg R, Hogrefe C, Kinney P, Knowlton K, Lynn B, Rosenthal J, Rosenzweig C, Patz J, Climate change, ambient ozone, and health in 50 US cities. *Clim Change*. 2007; 82(1–2):61–76. doi:10.1007/s10584-006-9166-7

Beniston, M.: The 2003 heat wave in Europe: A shape of things to come? An analysis based on Swiss climatological data and model simulations, *Geophys. Res. Lett*., 31, L02202, doi: 10.1029/2003GL018857, 2004.

Bloomer, B.J., J. W. Stehr, C. A. Piety, R. J. Salawitch, and R. R. Dickerson (2009), Observed relationships of ozone air pollution with temperature and emissions, *Geophys. Res. Lett*., *36,* L09803, doi:10.1029/2009GL037308.

Brauer, M., Amann, M., Burnett, R. T., et al (2011) Exposure assessment for estimation of the global burden of disease attributable to outdoor air pollution, *Environmental Science & Technology* 46, 652-660.

Butchart, N., and A. A. Scaife (2001), Removal of chlorofluorocarbons by increased mass exchange between the stratosphere and troposphere in a changing climate*, Nature,* 410, 799–802, doi:10.1038/35071047.

Christoudias, T., Pozzer, A., and Lelieveld, J.: Influence of the North Atlantic Oscillation on air pollution transport, *Atmos. Chem. Phys.*, 12, 869–877, doi:10.5194/acp-12-869-2012, 2012

Cionni, I., Eyring, V., Lamarque, J. F., Randel, W. J., Stevenson, D. S., Wu, F., Bodeker, G. E., Shepherd, T. G., Shindell, D. T., and Waugh, D. W.: Ozone database in support of CMIP5 simulations: results and corresponding radiative forcing, Atmos. Chem. Phys., 11, 11267-11292, doi:10.5194/acp-11-11267-2011, 2011.

Coleman, L., Martin, D., Varghese, S., Jennings, S. G., O'Dowd, C. D., 2013. Assessment of changing meteorology and emissions on air quality using a regional climate model: Impact on ozone. *Atmospheric Environment* 69, 198-210.

Colette, A., Bessagnet, B., Vautard, R., et al (2013) European atmosphere in 2050, a regional air quality and climate perspective under CMIP5 scenarios, Atmospheric Chemistry and Physics 13, 7451-7471, http://www.atmos-chem-phys.net/13/7451/2013/acp-13-7451-2013.pdf.

Collins, W. J., R. G. Derwent, B. Gamier, C. E. Johnson, M. G. Sanderson, and D. S. Stevenson (2003), Effect of stratosphere-troposphere exchange on the future tropospheric ozone trend, *J. Geophys. Res.,* 108(D12), 8528, doi:10.1029/2002JD002617.

COMEAP (2010) The mortality effects of long-term exposure to particulate air pollution in the United Kingdom, UK Department of Health Committee on the Medical Effects of Air Pollution. ISBN 978-0-85951-685-3, http://comeap.org.uk/documents/reports.html.

Cox, W.M., and Chu, S.-H., Assessment of interannual ozone variation in urban areas from a climatological perspective. *Atmos. Environ*. 30, 2615–2625, 1995.

Creilson, J. K., Fishman, J., and Wozniak, A. E.: Intercontinental transport of tropospheric ozone: a study of its seasonal variability across the North Atlantic utilizing tropospheric ozone residuals and its relationship to the North Atlantic Oscillation, *Atmos. Chem. Phys.,* 3, 2053–2066, doi:10.5194/acp-3-2053-2003, 2003.

Dawson, J. P., Adams, P. J., and Pandis, S. N.: Sensitivity of $PM_{2.5}$ to climate in the Eastern US: a modeling case study, *Atmos. Chem. Phys.,* 7, 4295-4309, doi:10.5194/acp-7-4295-2007, 2007.

Defra (2013) Air Pollution in the UK 2012, Department for Environment, Food and Rural Affairs, London. http://uk-air.defra.gov.uk/library/annualreport/air_pollution_uk_2012_issue_1.pdf.

Dentener, F., et al., The impact of air pollutant and methane emission controls on tropospheric ozone and radiative forcing: CTM calculations for the period 1990–2030, Atmos. Chem. Phys., 5, 1731–1755, 2005.

Dentener, F., et al., 2006: The global atmospheric environment for the next generation. *Environ. Sci. Technol*., 40, 3586-3594, doi:10.1021/es0523845.

Doherty, R. M., et al. (2013), Impacts of climate change on surface ozone and intercontinental ozone pollution: A multi-model study, *J. Geophys. Res. Atmos.*, 118, doi:10.1002/jgrd.50266.

Dominici, F., Peng, R. D., Barr, C. D. and Bell, M. L. (2010) Protecting human health from air pollution shifting from a single-pollutant to a multipollutant approach, *Epidemiology* 21, 187-194.

Duncan B. N. and Bey, I.: A modeling study of the export pathways of pollution from Europe: Seasonal and interannual variations (1987–1997), *J. Geophys. Res.-Atmos.*, 109, D08301,doi:10.1029/2003JD004079, 2004

EAC (2010) Air Quality. House of Commons Environmental Audit Committee. Fifth report of session 2009-10., Report no. HC 229-I. The Stationary Office Ltd., London. www.publications.parliament.uk/pa/cm200910/cmselect/cmenvaud/229/229i.pdf.

Eckhardt, S., Stohl, A., Beirle, S., Spichtinger, N., James, P., Forster, C., Junker, C., Wagner, T., Platt, U., and Jennings, S. G.: The North Atlantic Oscillation controls air pollution transport to the Arctic, *Atmos. Chem. Phys*., 3, 1769-1778, doi:10.5194/acp-3-1769-2003, 2003.

EEA (2005) Environment and Health. EEA Report No 10/2005., European Environment Agency. ISBN: 92-9167-781-7. http://www.eea.europa.eu/publications/eea_report_2005_10.

EEA (2014) Air quality in Europe - 2014 report. EEA Report No 5/2014., European Environment Agency. ISBN: 978-92-9213-490-7. http://www.eea.europa.eu/publications/air-quality-in-europe-2014.

Emberson, L. D., Kitwiroon, N., Beevers, S., Büker, P., and Cinderby, S.: Scorched Earth: how will changes in the strength of the vegetation sink to ozone deposition affect human health and ecosystems?, *Atmos. Chem. Phys*., 13, 6741-6755, doi:10.5194/acp-13-6741-2013, 2013.

Fang Y, Mauzerall DJ, Liu J, Fiore AM, Horowitz, Impacts of 21st century climate change on global air pollution-related premature mortalityClimatic Change. 2013; 121:239–253, doi: 10.1007/s10584-013-0847-8

Filleul, L., Cassadou, S., Medina, S., Fabres, P., Lefranc, A., Eilstein, D., Le Tertre, A., Pascal, L., Chardon, B., Blanchard, M.,Declercq, C., Jusot, J.-F., Prouvost, H., and Ledrans, M.: The Relation Between Temperature, Ozone, and Mortality in Nine French Cities During the Heat Wave of 2003, Environ. Health Perspect., 114, 1344–1347, doi: 10.1289/ehp.8328, 2006.

Fiore, A. M., Naik, V., Spracklen, D. V., et al (2012) Global air quality and climate, *Chemical Society Reviews* 41, 6663-6683.

Fischer, P. H., B. Brunekreef, and E. Lebret (2003), Air pollution related deaths during the 2003 heat wave in the Netherlands, *Atmos. Environ.*, 38, 1083-1085.

Forkel, R. and R. Knoche, 2006: Regional climate change and its impact on photooxidant concentrations in southern Germany: Simulations with a coupled regional climate-chemistry model. *J. Geophys. Res*., 2006, 111, D12302.

Francis, X.V., C. Chemel, R.S. Sokhi, E.G. Norton, H.M.A. Ricketts, B.E.A. Fisher, (2011) Mechanisms responsible for the build-up of ozone over South East England during the August 2003 heatwave, *Atmospheric Environment,* 45, 38, 6880-6890

Glotfelty, T., Zhang, Y., Karamchandani, P. and Streets, D. G. (2013) Will the role of intercontinental transport change in a changing climate?, *Atmospheric Chemistry and Physics*, 14, 9379-9402.

Hauglustaine, D.A., Lathiere, J., Szopa, S., Folberth, G.A., 2005. Future tropospheric ozone simulated with a climate-chemistry-biosphere model. *Geophys. Res. Lett. 32*, L24807, doi:10.1029/2005GL024031.

Heal, M. R., Heaviside, C., Doherty, R. M., Vieno, M., Stevenson, D. S. and Vardoulakis, S. (2013) Health burdens of surface ozone in the UK for a range of future scenarios, *Environment International* 61, 36-42.

Hedegaard, G. B., Brandt, J., Christensen, J. H., Frohn, L. M., Geels, C., Hansen, K. M., and Stendel, M.: Impacts of climate change on air pollution levels in the Northern Hemisphere with special focus on Europe and the Arctic, Atmos. Chem. Phys., 8, 3337-3367, doi:10.5194/acp-8-3337-2008, 2008.

Held, Isaac M., Brian J. Soden, 2006: Robust Responses of the Hydrological Cycle to Global Warming. *J. Climate*, **19**, 5686–5699. doi: http://dx.doi.org/10.1175/JCLI3990.1

Hodzic, A., Madronich, S., Bohn, B. Massie, S., Menut, L., andWiedinmyer, C.: Wildfire particulate matter in Europe during summer 2003: meso-scale modeling of smoke emissions, transport and radiative effect., *Atmos. Chem. Phys.*, 7, 4043–4064, 2007

Health Protection Agency, 2012: Health Effects of Climate Change in the UK 2012 – Current evidence, public health recommendations and research gaps. Health Protection Agency. Centre for Radiation, Chemical and Environmental Hazards, UK. Vardoulakis, S. and Heaviside, C. [eds.] ISBN: 978-0-85951-723-2.

Hurrell, J.W., 1995: Decadal trends in the North Atlantic Oscillation and relationships to regional temperature and precipitation. *Science* 269, 676-679.

IPCC (2001) Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change
[Houghton, J.T., et al., (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC (2007) Climate Change 2013: The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, 2007, Solomon, S., et al., (eds.) Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC (2013), Climate Change 2013: The Physical Science Basis, Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker,T.F., et al., (eds.)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Isaksen, I.S.A., et al., 2009: Atmospheric composition change: Climate-chemistry interactions. *Atmos. Environ*., 43, 5138-5192, doi:10.1016/j.atmosenv.2009.08.003.

Jacob, D. J.,1999, Introduction to Atmospheric Chemistry, Princeton University Press, Princeton, USA: http://acmg.seas.harvard.edu/people/faculty/djj/book/

Jacob, D. J. and Winner, D. A. (2009) Effect of climate change on air quality, *Atmospheric Environment* 43, 51-63.

Jacobson, M. Z., 2002, Atmospheric Pollution: History, Science and Regulation, Cambridge Universty Press, Cambridge and USA, http://web.stanford.edu/group/efmh/jacobson/POLbook/POLbook.html

Jaeger, C. C., Krause, J., Haas, A., Klein, R. and Hasselmann, K. (2008) A method for computing the fraction of attributable risk related to climate damages, *Risk Analysis* 28, 815-823.

Jerrett M et al. (2009a). Long- term ozone exposure and mortality. *New England Journal of Medicine*, 360(11):1085–1095.

Johnson, C.E., W.J. Collins, D.S. Stevenson, R.G. Derwent (1999), Relative roles of climate and emissions changes on future oxidant concentrations, *J. Geophys. Res*., 104, D15, 18631-18645, doi:10.1029/1999JD900204.

Johnson, H., Kovats, S., McGregor, G., Stedman, J., Gibbs, M., Walton, H., 2005. The impact of the 2003 heat wave on daily mortality in England and Wales and the use of rapid weekly mortality estimates. *Euro Surveill.* 10 (7), 558.

Katragkou, E., P. Zanis, I. Kioutsioukis, I. Tegoulias, D. Melas, B. C. Krüger, and E. Coppola (2011), Future climate change impacts on summer surface ozone from regional climate-air quality simulations over Europe, *J. Geophys. Res*., 116, D22307, doi:10.1029/2011JD015899.

Katsouyanni K, Samet JM, Anderson HR, Atkinson RW, Le Tertre A, Medina S, et al. 2009. Air Pollution and Health: A European and North American Approach (APHENA). Research Report 142. Boston:Health Effects Institute. Available: http://pubs.healtheffects.org/view.php?id=327

Kim, M. J., Park, R. J., Ho, C. H., Woo, J. H., Choi, K. C., Song, C. K., Lee, J. B., 2015. Future ozone and oxidants change under the RCP scenarios. *Atmospheric Environment* 101, 103-115.

Kirtman, B., S.B. Power, et al., 2013: Near-term Climate Change: Projections and Predictability. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker,T.F., et al., (eds.)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Knowlton K, Rosenthal JE, Hogrefe C, Lynn B, Gaffin S, Goldberg R, et al. Assessing ozone-related health impacts under a changing climate. *Environ Health Perspect* 2004; 112:1557–63.

Krewski D et al. (2009). Extended follow-up and spatial analysis of the American Cancer Society study linking particulate air pollution and mortality. Boston, Health Effects Institute (Research Report 140; http://pubs.healtheffects.org/getfile.php?u=478

Kunkel, K. E., H. C. Huang, X. Z. Liang, J. T. Lin, D. Wuebbles, Z. Tao, A. Williams, M. Caughey, J. Zhu, and K. Hayhoe (2008), Sensitivity of future ozone concentrations in the Northeast U.S. to U.S. regional climate change, Mitigation and Adaptation Strategies for Global Change, 13, 597–606, doi:10.1007/s11027-007-9137-y.

Lacressonnière, G., Peuch, V. H., Vautard, R., Arteta, J., Deque, M., Joly, M., Josse, B., Marcal, V., Saint-Martin, D., 2014. European air quality in the 2030s and 2050s: Impacts of global and regional emission trends and of climate change. *Atmospheric Environment* 92, 348-358

Lamarque, J.-F., et al., Historical (1850-2000) gridded anthropogenic and biomass burning emissions of reactive gases and aerosols: methodology and application, *Atmos. Chem. Phys*., 10, 7017–7039, 2010.

Langner, J., Engardt, M. and Andersson, C. (2012a) European summer surface ozone 1990-2100, *Atmospheric Chemistry and Physics* 12, 10097-10105.

Langner, J., Engardt, M., Baklanov, A., et al (2012b) A multi-model study of impacts of climate change on surface ozone in Europe, *Atmospheric Chemistry and Physics* 12, 10423-10440.

Lee, J. D., Lewis, A. C., Monks, P. S., Jacob, M., Hamilton, J. F., Hopkins, J. R., Watson, N. M., Saxton, J. E., Ennis, C., Carpenter, L. J., Carslaw, N., Fleming, Z., Bandy, B. J., Oram, D. E., Penkett, S. A., Slemr, J., Norton, E., Rickard, A. R., Whalley, L. K., Heard, D. E., Bloss, W. J., Gravestock, T., Smith, S. C., Stanton, J., Pilling, M. J., and Jenkin, M. E.: Ozone photochemistry and elevated isoprene during the UK heatwave of August 2003, *Atmos. Environ*., 40, 7598–7613, 2006

Li, Q. B., Jacob, D. J., Bey, I., Palmer, P. I., Duncan, B. N., Field, B. D., Martin, R. V., Fiore, A.M., Yantosca, R. M., Parrish, D. D., Simmonds, P. G., and Oltmans, S. J.: Transatlantic transport of pollution and its effects on surface ozone in Europe and North America, *J. Geophys. Res*., 107, 4166, doi:10.1029/2001JD001422, 2002

Likhvar, V. N., Pascal, M., Markakis, K., Colette, A., Hauglustaine, D., Valari, M., Klimont, Z., Medina, S., Kinney, P., 2015. A multi-scale health impact assessment of air pollution over the 21st century. *Science of the Total Environment* 514, 439-449.

Lim, S. S., Vos, T., Flaxman, A. D., et al (2012) A comparative risk assessment of burden of disease and injury attributable to 67 risk factors and risk factor clusters in 21 regions, 1990-2010: a systematic analysis for the Global Burden of Disease Study 2010, *The Lancet* 380, 2224-2260.

Liu C., Allan, R.P., 2013 Observed and simulated precipitation responses in wet and dry regions 1850–2100, *Environ. Res. Lett.* **8** 034002

Markakis, K., Valari, M., Colette, A., Sanchez, O., Perrussel, O., Honore, C., Vautard, R., Klimont, Z., Rao, S., 2014. Air quality in the mid-21st century for the city of Paris under two climate scenarios; from the regional to local scale. *Atmospheric Chemistry and Physics* 14, 7323-7340.

Masato, G., B. J. Hoskins and T. J. Woollings (2013). Winter and summer Northern Hemisphere blocking in CMIP5 models. *J. Climate,* 26, 7044–7059. doi: http://dx.doi.org/10.1175/JCLI-D-12-00466.1

Masui, T., et al., An emission pathway for stabilization at 6 Wm-2 radiative forcing, *Climatic Change*, 109, 59-76, 2011.

Mickley, L.J., D.J. Jacob, B.D. Field, and D. Rind, 2004: Effects of future climate change on regional air pollution episodes in the United States. *Geophys. Res. Lett.*, **31**, L24103, doi:10.1029/2004GL021216.

Meleux, F., F. Solmon, and F. Giorgi (2007), Increase in summer European ozone amounts due to climate change, *Atmos. Environ*., 41(35), 7577–7587, doi:10.1016/j.atmosenv.2007.05.048.

Moxim, W.J., et al., Simulated global tropospheric PAN: its transport and impact on NOx, *J. Geophys. Res*., 101, 12621-12638, 1996

Murazaki, K. and P. Hess (2006). How does climate change contribute to surface ozone change over the United States? *J. Geophys. Res. 111,* D05301, doi:10.1029/2005JD005873.

Nakicenovic, N., et al., Special Report on Emission Scenarios (SRES) Report of Working Group III of the Intergovernmental Panel on Change, Cambridge University Press, Cambridge, U.K., 2000.

O'Connor, F. M., et al. (2010), Possible role of wetlands, permafrost, and methane hydrates in the methane cycle under future climate change: A review, *Rev. Geophys*., 48, RG4005, doi:10.1029/2010RG000326.

Orru H, Andersson C, Ebi K L, Langner J, Astrom C and Forsberg B, Impact of climate change on ozone-related mortality and morbidity in Europe, 2013 *Eur. Respir.* J. 41 285–94, doi: 10.1183/09031936.00210411.

Patz, J. A., Frumkin, H., Holloway, T., Vimont, D. J., Haines, A., 2014. Climate Change Challenges and Opportunities for Global Health. Jama-Journal of the American Medical Association 312, 1565-1580.

Pattenden, S., Armstrong, B. G., Milojevic, A., Heal, M. R., Chalabi, Z., Doherty, R., Barratt, B., Kovats, R. S. and Wilkinson, P. (2010) Ozone, heat and mortality: acute effects in 15 British conurbations, *Occupational & Environmental Medicine* 67, 699-707.

Pausata, F. S. R., Pozzoli, L., Vignati, E., and Dentener, F. J.: North Atlantic Oscillation and tropospheric ozone variability in Europe: model analysis and measurements intercomparison, *Atmos. Chem. Phys*., 12, 6357-6376, doi:10.5194/acp-12-6357-2012, 2012.

Penrod, A., Zhang, Y., Wang, K., Wu, S. Y., Leung, L., 2014. Impacts of future climate and emission changes on US air quality. *Atmospheric Environment* 89, 533-547.

Pope, C. A., Ezzati, M. and Dockery, D. W. (2009) Fine-particulate air pollution and life expectancy in the United States, *The New England Journal of Medicine* 360, 376-386.

Pye, H.O.T, H. Liao, S. Wu, L. J. Mickley, D. J. Jacob, D. K. Henze and J. H. Seinfeld, *J. Geophys. Res*., 2009,114, D01205.

Rasmussen, D., Hu, J., Mahmud, A., Kleeman, M. J., 2013. The Ozone-Climate Penalty: Past, Present, and Future. *Environmental Science & Technology* 47, 14258-14266.

Rao, S., Chirkov, V., Dentener, F., Van Dingenen, R., Pachauri, S., Purohit, P., Amann, M., Heyes, C., Kinney, P., Kolp, P., Klimont, Z., Riahi, K., and Schoepp, W. (2012).Environmental Modeling and Methods for Estimation of the Global Health Impacts of Air Pollution, *Environ. Model. Assess.*, doi:10.1007/s10666-012-9317-3, 2012

Racherla, P.N., and P.J. Adams (2008), The response of surface ozone to climate change over the Eastern United States. *Atmos. Chem. Phys.*, *8,* 871-885, doi:10.5194/acp-8-871-2008.

Racherla, P.A., and P. J. Adams, *J. Geophys. Res.*, 2006,111, D24103.

Real, E., Law, K. S., Wienzierl, B., Fiebig, M., Petzold, A., Wild, O., Methven, J., Arnold, S., Stohl, A., Huntrieser, H., Roiger, A., Schlager, H., Stewart, D., Avery, M., Sachse, G., Brow- ell, E., Ferrare, R., and Blake, D.: Processes influencing ozone levels in Alaskan forest fires plumes during long-range transport over the North Atlantic, *J. Geophys. Res*., 112, D10S41, doi:10.1029/2006JD007 576, 2007.

Real, E., Orlandi, E., Law, K. S., Fierli, F., Josset, D., Cairo, F., Schlager, H., Borrmann, S., Kunkel, D., Volk, C. M., McQuaid, J. B., Stewart, D. J., Lee, J., Lewis, A. C., Hopkins, J. R., Ravegnani, F., Ulanovski, A., and Liousse, C.: Cross-hemispheric transport of central African biomass burning pollutants: implications for downwind ozone production, *Atmos. Chem. Phys.,* 10, 3027-3046, doi:10.5194/acp-10-3027-2010, 2010.

Ren C, Williams GM, Mengersen K, Morawska L, Tong S. Does temperature modify short-term effects of ozone on total mortality in 60 large eastern US communities?—An assessment using the NMMAPS data. *Environ Int* 2008;34:451–8.

Riahi, K., et al., RCP 8.5-A scenario of comparatively high greenhouse gas emissions, *Climatic Change,* 109, 33-57, 2011.

Riahi, K., Dentener, F., Gielen, D., Grubler, A., Jewell, J., Klimont, Z., Krey, V., McCollum, D., Pachauri, S., Rao, S., van Ruijven, B., van Vuuren, D. P., and Wilson, C.: Energy Pathways for Sustainable Development, in: Global Energy Assessment: Toward a Sustainable Future, edited by: Nakicenovic, N., IIASA, Laxenburg, Austria and Cambridge University Press, Cambridge, UK and New York, NY, USA, 2012

Royal Society (2008) Ground-level ozone in the 21st century: future trends, impacts and policy implications. Science Policy Report 15/08., The Royal Society, London. ISBN: 978-0-85403-713-1. http://royalsociety.org/policy/publications/2008/ground-level-ozone/.

Savage, N. H.,et al.: The sensitivity of Western European NO2 columns to interannual variability of meteorology and emissions: a model– GOME study, *Atmos. Sci. Lett.,* 9(4), 182–188, doi:10.1002/asl.193, 2008

Scḧar, C., Vidale, P. L., L̈uthi, D., Frei, C., Ḧaberli, C., Liniger, M. A., and Appenzeller, C.: The role of increasing temperature variability in European summer heatwaves, *Nature,* 427, 332–336, 2004

Shindell, D., J.C.I. Kuylenstierna, E. Vignati, R. van Dingenen, M. Amann, Z. Klimont, S.C. Anenberg, N. Muller, G. Janssens-Maenhout, F. Raes, J. Schwartz, G. Faluvegi, L. Pozzoli, K. Kupiainen, L. Höglund-Isaksson, L. Emberson, D. Streets, V. Ramanathan, K. Hicks, N.T.K. Oanh, G. Milly, M. Williams, V. Demkine, and D. Fowler, 2012: Simultaneously mitigating near-term climate change and improving human health and food security. *Science*, 335, 183-189, doi:10.1126/science.1210026.

Sillman, S. and P. J. Samson (1995), The impact of temperature on oxidant formation in urban, polluted rural and remote environments, *J. Geophys. Res., 100,* D7, 11497–11508, doi:10.1029/94JD02953.

Silva, R.A., et al., Global premature mortality due to anthropogenic outdoor air pollution and the contribution of past climate change, *Environ. Res. Lett*. 8 034005 doi:10.1088/1748-9326/8/3/034005

Smith KR, Jerrett M, Anderson HR, Burnett RT, Stone V, Derwent R, et al. 2009. Public health benefits of strategies to reduce greenhouse-gas emissions: health implications of short-lived greenhouse pollutants. *Lancet* 374:2091–2103.

Solberg, S., Hov, O., Sovde, A., Isaksen, I. S. A., Coddeville, P., De Backer, H., Forster, C., Orsolini, Y., and Uhse, K.: European surface ozone in the extreme summer 2003, J. *Geophys. Res.Atmos*., 113, D07307, doi:10.1029/2007JD009098, 2008.

Squire, O. J., Archibald, A. T., Abraham, N. L., Beerling, D. J., Hewitt, C. N., Lathi+¿re, J., Pike, R. C., Telford, P. J., Pyle, J. A., 2014. Influence of future climate and cropland expansion on isoprene emissions and tropospheric ozone. *Atmospheric Chemistry and Physics* 14, 1011-1024.

Stedman, J. R. (2004) The predicted number of air pollution related deaths in the UK during the August 2003 heatwave, *Atmospheric Environment* 38, 1087-1090.

Stevenson, D.S., F. Dentener et al., Multi-model ensemble simulations of present-day and near-future tropospheric ozone, *J. Geophys. Res*., 111, D08301, doi:10.1029/2005JD006338, 2006.

Stock, Z. S., Russo, M. R., Pyle, J. A., 2014. Representing ozone extremes in European megacities: the importance of resolution in a global chemistry climate model. *Atmospheric Chemistry and Physics* 14, 3899-3912.

Stott, P. A., Stone, D. A. and Allen, M. R. (2004) Human contribution to the European heatwave of 2003, *Nature* 432, 610-614.

Tagaris E, Liao K-J, DeLucia AJ, Deck L, Amar P, Russell AG. Potential impact of climate change on air pollution-related human health effects. *Environ Sci Technol*. 2009; 43(13):4979–4988. doi:10.1021/es803650w

Task Force on Hemispheric Transport of Air Pollution (TF-HTAP) (2011), *Hemispheric Transport of Air Pollution 2010, Part A: Ozone and Particulate Matter, Air Pollut. Stud., 17,* edited by: Dentener, F., Keating, T. and Akimoto, H., U. N. Econ. Comm. Eur., Geneva, Switzerland, available at: http://www.htap.org.

Thomson, A.M., et al., RCP4.5: a pathway for stabilization of radiative forcing by 2100, *Climatic Change,* 109, 77-94, 2011.

Tressol, M., Ordonez, C., Zbinden, R., Brioude, J., Thouret, V., Mari, C., Nedelec, P., Cammas, J.-P., Smit, H., Patz, H.-W., and Volz-Thomas, A.: Air pollution during the 2003 European heat wave as seen by MOZAIC airliners, *Atmos. Chem. Phys*., 8, 2133-2150, doi:10.5194/acp-8-2133-2008, 2008.

Uno, I.,et al.,: Systematic analysis of interannual and seasonal variations of model-simulated tropospheric NO2 in Asia and comparison with GOME-satellite data, Atmos. Chem. Phys., 7, 1671–1681, 2007, http://www.atmos-chem-phys.net/7/1671/2007/

van Vuuren, D.P., et al., RCP2.6: exploring the possibility to keep global mean temperature increase below 2 degrees C, *Climatic Change*, 109, 95-116, 2011.

Vautard, R., Honoré, C., Beekmann, M., Rouil, L., 2005. Simulation of ozone during the August 2003 heat wave and emission control scenarios. *Atmos. Environ*. 39,2957-2967.

Vieno, M., Dore, A. J., Stevenson, D. S., Doherty, R., Heal, M. R., Reis, S., Hallsworth, S., Tarrason, L., Wind, P., Fowler, D., Simpson, D., and Sutton, M. A.: Modelling surface ozone during the 2003 heat-wave in the UK, *Atmos. Chem. Phys.*, 10, 7963-7978, doi:10.5194/acp-10-7963-2010, 2010.

UNEP/WMO, (2011) UNEP/WMO Integrated Assessment of Black Carbon and Tropospheric Ozone (2011). http://www.unep.org/dewa/Portals/67/pdf/BlackCarbon_report.pdf

WHO, (2006) Air quality guidelines. Global update 2005. Particulate matter, ozone, nitrogen dioxide and sulfur dioxide., World Health Organisation Regional Office for Europe, Copenhagen. ISBN 92 890 2192 6. http://www.euro.who.int/__data/assets/pdf_file/0005/78638/E90038.pdf

WHO (2013) Review of evidence on health aspects of air pollution – REVIHAAP Project: Technical Report, World Health Organisation, Copenhagen, Denmark. http://www.euro.who.int/__data/assets/pdf_file/0004/193108/REVIHAAP-Final-technical-report-final-version.pdf.

Wu, S., Mickley, L. J., Jacob, D. J., Rind, D. and Streets, D. G. (2008a) Effects of 2000-2050 changes in climate and emissions on global tropospheric ozone and the policy-relevant background surface ozone in the United States, Journal of Geophysical Research 113, doi:10.1029/2007JD009639.

Wu, S., Mickley, L. J., Leibensperger, E. M., Jacob, D. J., Rind, D. and Streets, D. G. (2008b) Effects of 2000-2050 global change on ozone air quality in the United States, *Journal of Geophysical Research* 113, doi:10.1029/2007JD008917.

Young, P. J.,et al., Pre-industrial to end 21st century projections of tropospheric ozone from the Atmospheric Chemistry and Climate Model Intercomparison Project (ACCMIP), *Atmos. Chem. Phys.*, 13, 2063-2090, doi:10.5194/acp-13-2063-2013, 2013.

Zeng, G., J. A. Pyle, and P. J. Young, P. J.: Impact of climate change on tropospheric ozone and its global budgets, *Atmos. Chem. Phys., 8,* 369–387, 2008.