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Assessment Report on the link and feedbacks between climate change and air pollution including wildfires – A European perspective

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Introduction

This assessment report is the first scientific deliverable from Work Package 3 of the European H2020 EXHAUSTION project. The EXHAUSTION project aims to quantify the changes in cardiopulmonary mortality and morbidity due to extreme heat and air pollution (including from wildfires) under selected climate scenarios while including a diverse set of adaptation mechanisms and strategies, calculate the associated costs, and identify effective strategies for minimizing adverse impacts. The results will be disseminated to the general public and key decision- and policy-makers across Europe, providing a tool to increase European resilience towards climate change.

A specific objective of the project is to provide quantitative projections of occurrence and population exposure to extreme heat and air pollution (particulate matter and ozone) in Europe for 2030, 2050, and 2100 under different emission scenarios. Three chemistry-transport models (CTMs): the Danish Eulerian Hemispheric Model (DEHM), the System for Integrated Modelling of Atmospheric Composition (SILAM) and Weather Research Forecast model with Chemistry (WRF-Chem), coupled to the Community Earth System Model (CESM), will be used in order to perform high-resolution simulations of current and future air pollution over Europe. The current report will assess the state-of-the-art in terms of modeling and current knowledge on the link and feedbacks between climate change and air pollution, including wildland fires.

Summary

This assessment report builds on recent research on the interactions between climate change and air pollution, highlighting projections on air pollution under different future climate scenarios and time horizons, and is based on a literature review of research articles and reports from the period 2010-2020. The assessment focuses on:

(i) the isolated impacts of climate change on air pollution and natural emissions focusing on Europe and in near future (2030), by mid-century (2050) and by end of the century (2100),

(ii) impacts of the combined changes in air pollution and emissions on climate,

(iii) impacts of climate change on wildland fires and air pollutant levels,

(iv) the role of adaptation and mitigation policies on climate change and air pollution.

Climate change can affect air pollution both physically, by modifying transport and mixing of pollutants as well as emissions of natural origin, and chemically by affecting chemical reactions. Such changes can lead to the so-called “climate change penalty” which can be defined as the deterioration of air quality due to a warming climate, in the absence of changes in anthropogenic polluting activities (Fu and Tian, 2019). Changes in climate also influence the frequency and intensity of natural wildland fires. Vice versa, pollutants in the atmosphere and their precursor emissions can influence climate by modifying clouds properties, precipitation, and radiation. Feedbacks between climate change and air pollution should be taken into account in developing mitigation/adaptation strategies. However, the existing knowledge gaps on the involved mechanisms result in large uncertainties in the feedbacks between climate change and air pollution. For instance, although the main cause of wildfires currently is anthropogenic, i.e. caused by agriculture and forest management practices, projections suggest that the risk of wildland fires will increase.

Secondary pollutants like ozone or organic aerosols are expected to increase with the projected warming, mainly due to an increase in the oxidation capacity of the atmosphere and the increase in natural emissions. On the other hand, the projected increases in precipitation are expected to enhance removal of particles from the atmosphere, resulting in a weak positive or negative change. It is important to highlight that these changes vary geographically and seasonally. It is, however, important to note that all studies reviewed here show that changes in anthropogenic emissions have much larger impacts on air pollutant concentrations compared to changes in climate, and that there are compensating effects of changes in emissions and climate, as summarized above. This strongly suggests that feedbacks between pollutants and climate change play a crucial role in the direction of the response of future climate change and air pollution.

Studies show that there may be immediate benefits for air quality from mitigation of greenhouse gas emissions (GHG) due to reduced emissions of co-emitted air pollutants, such as the aerosols (denoted “air quality co-benefits”). Noteworthy, the reverse relation does not hold: improving air quality often raises the challenges to the climate change mitigation. The improvement in air quality depends on, *i.e.,* the sources of GHG being mitigated and the amount of co-emitted species, and finally the GHG mitigation measures being taken. Socioeconomic and technological developments may influence the potential for air quality co-benefits. Recent findings suggest that limiting climate change mitigation today would entail higher air pollution levels in the future. However, emission controls should continue to be imposed irrespective of climate policy seen in light of the adverse health impacts of these pollutants, while these reductions in aerosols could lead to an increase in warming due to decreased negative radiative forcing via these aerosols.

The main conclusion from this review is that the estimated changes in pollutant levels in the future vary significantly depending on the model systems and the scenarios used in the different studies. Nevertheless, studies generally agree on the tendency of the changes in pollutant levels due to climate change.

The assessment report consists of an introduction (section 1), where the different physical and chemical processes in relation to climate change and air pollution are introduced; sections 2 and 3 summarize the recent findings on the isolated impacts of climate change on ozone and particulate matter concentrations, respectively; section 4 compares the relative impact of future emissions on air quality versus the impacts on air quality caused by climate change alone; section 5 summarizes recent findings on the impacts of atmospheric composition on climate change; section 6 summarizes the impacts of climate change on wildland fires and consequently on air pollution and climate change; section 7 summarizes the co-benefits of climate adaptation and mitigation on air pollution; section 8 introduces examples of advanced extreme weather and wildland fire warning systems as a tool for mitigation and adaption; and finally, section 9 offers some recommendations regarding needs for future research.

1. Introduction to interaction mechanisms

Meteorology is a major driver of the fate of all chemically active species within the atmosphere, including climate-relevant compounds like carbon dioxide (CO2), methane (CH4), ozone (O3) and secondary inorganic (e.g. sulfate and nitrate) and organic aerosols. Changes in meteorological factors can affect atmospheric composition in several ways: (i) Changes in temperatures, precipitation and wind patterns may affect the natural sources of aerosols in the atmosphere, including mineral dust, sea salt, and biogenic volatile organic compounds (BVOCs). (ii) Changes in climate parameters can affect the anthropogenic sources. E.g., increasing temperatures will increase NH3 emissions from agricultural activities and VOC emissions from loading of petroleum products. (iii) Changes in temperature, humidity, solar radiation, and other meteorological factors can affect the rate of photochemical, oxidative and other chemical reactions, speeding up or slowing down the chemical transformation of emitted species. (iv) Changes in wind patterns and precipitation affect transport and mixing of atmospheric components, as well as deposition processes. (v) Changes in large-scale weather patterns, which control the dispersion and transport of air pollutants, may result in changes of the frequency and intensity of air pollution events. In summary, to understand how climate change may affect air pollution, one needs to consider how climate affects the full range of biological, chemical and physical processes. Figure 1 summarizes how climate affects air quality and the two-way interactions between emissions and climate.

As discussed in Fiore et al. (2019), there are several approaches that are used in the literature in order to understand how O3 and particulate matter (PM) are influenced by changes in climate or emissions. These include (i) sensitivity studies in which individual meteorological parameters are perturbed, (ii) statistical downscaling of future changes in meteorology, using correlations between observed changes in air quality and meteorological variables from climate models, (iii) the direct calculation of air quality by using fully coupled global or regional chemistry–climate models (CCM), global-to-urban CCMs, or off-line global or regional chemical transport models (CTMs) forced by either using projections of meteorological fields from separate atmosphere–ocean general circulation models, or dynamically downscaled meteorology.

1.1. Climate change impacts on natural emissions

Wind speed and sea-surface temperatures also modify the production of sea-salt aerosols (SSA) emissions (Soares et al., 2016). The two main mechanisms responsible for sea spray formation are air bubble bursting during whitecap formation, and decay and direct tearing of droplets from the top of breaking waves. Therefore, the formation of primary SSA is mainly dependent on wind speed (Monahan et al., 1986). In addition, laboratory (Mårtensson et al., 2003) and in situ measurements (Nilsson et al., 2007) showed that for nano-sized particles, the aerosol number emission in nano sizes decreases with increasing seawater temperature, while number emissions of particles larger than 100 nm increase with increasing seawater temperature.

Wind speed and precipitation also impacts the mineral dust emissions and their transport. Changes in wind can modify both the emissions of mineral dust particles, as well as their transport pathways. The dust emissions depend on the availability of fine crustal material that can be lifted from the ground when surface wind velocity exceeds a certain threshold. This threshold depends on surface roughness (e.g. rocks and vegetation), grain size and soil moisture. Mineral dust is an important natural source of atmospheric aerosols, and interfere with the atmosphere's radiation, impacting climate (Klingmuller et al., 2019). However, there are large variations among projections of the future global mineral dust burden

(Boucher et al., 2013). While some studies show decreases by up to 60% globally, mainly due to the effect of CO2 fertilization on vegetation (Mahowald et al., 2006), others project moderate (10-20%: Tegen et al., 2004; Jacobson and Streets, 2009; Liao et al., 2009) to large increases (tripling) due to large increases in bare soil fraction (Woodward et al., 2005). The large range reflects different responses of the atmosphere and vegetation cover to climate change, and results in low confidence in these predictions.

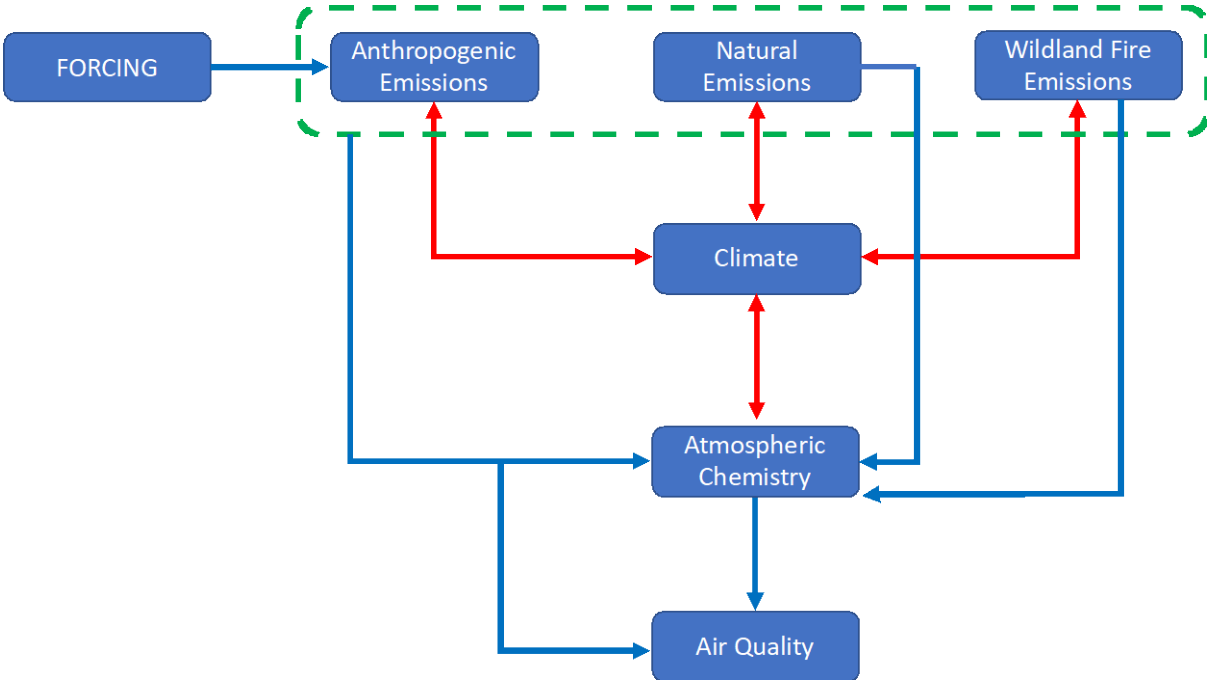

Figure 1. Interactions between climate, emissions and air quality (adopted from Jacob and Winner, 2009). The green dashed box shows the different emissions, while the red and the blue arrows show two-way and one-way interactions, respectively.

Increasing temperatures also cause larger emissions of biogenic VOCs such as isoprene and terpenes, which are emitted by the vegetation, and are important precursor for O3 and secondary organic aerosols (SOA) (Guenther et al., 2006). Recent studies also show that increasing CO2 concentrations limits biogenic emissions as the stomata closes with increased CO2 (Arneth et al., 2007). The presence of highly reactive biogenic VOCs such as isoprene is only slightly favoring aerosol yield or even tends to have a decreasing effect, while aerosol formation is enhanced in the presence of other compounds such as mono- or sesquiterpenes. In contrast, the importance of isoprene for O3 formation is higher than that of other terpenoids. In addition to the type of the biogenic species and the availability of nitrogen oxides as reaction partner, recent investigations show that positive feedbacks exist between biogenic and anthropogenic VOCs for aerosol formation as well as ozone formation (Grote, 2019).

1.2. Climate change impacts on anthropogenic emissions

Meteorological parameters like surface wetness, wind and especially temperature have also been shown to have an important impact on emissions related to human activities. The ammonia emissions (NH3) from agricultural sources have been projected to increase significantly in Europe in a future climate (Skjøth and Geels, 2013) and at the global scale an empirical estimate proposes that a 5^{o}C temperature

increase would increase NH3 emission by 42 % [28-67%] (Sutton et al., 2013). This suggested climate penalty on NH3 can through the conversion to ammonium (NH4+) also affect the PM2.5 levels in Europe. In addition, changes in temperatures also impact the use of ventilation systems, thus modifying energy production and related emissions; warmer temperatures increase the energy demand for cooling in the summer, while decreasing the demand for heating in the winter (EEA, 2019a). In addition, changes to the hydrological cycle have an impact on hydropower (EEA, 2019a). An important source for PM in Europe is residential wood combustion, e.g. small-scale combustion units, where the ambient temperature influences the extent of the use.

1.3. Climate change impacts on chemical processes

The role of meteorological factors such as temperature and relative humidity for atmospheric chemistry is to a large extent through its influence on the hydroxyl radical (OH), which is the major component for the oxidizing capacity of the troposphere and thus crucial for the chemical removal of pollutants. The primary sources of OH are the photolysis of O3, and reaction with water vapor, the latter increasing with a warming climate (Seinfeld and Pandis, 2016). Increase in temperatures results in an increase in summertime ozone over polluted areas due to increased photochemical activity, and a reduction of background ozone due to an expected increase in water vapor (Jacob and Winner, 2009). In addition, increasing temperatures also increase sulfate aerosol (SO42-) formation due to faster sulfur dioxide (SO2) oxidation, while nitrate (NO3-) particles decrease as ammonium nitrate evaporates and nitrate shifts from the particle phase to the gas phase (Dawson et al., 2007). On the other hand, increasing temperatures can also reduce SO4= aerosols due to less available OH to react with SO2 in order to produce SO42-, as increased biogenic emissions also react with the available OH, thus competing with SO2 (Im et al., 2012). SOA can both be reduced due to the shift from particle to gas phase, and increase due to the increase of biogenic SOA precursors (Sheehan and Bowman, 2001; Kanakidou et al., 2005; Tsigaridis and Kanakidou, 2007).

1.4. Climate change impacts on production, dispersal and removal rates

In addition to temperature, there are other meteorological factors affecting the production, dispersion, and removal of air pollutants. Climate change can modify the large-scale circulations (Otero et al., 2018), thus ventilation (wind speed/direction, mixing height). Lower wind speeds in polluted regions can cause O3 concentrations to increase due to longer residence/reaction time and due to increased aerodynamic resistance to dry deposition (Dawson et al., 2007). Mixing heights have a more complicated effect on O3 concentrations. Increasing mixing height increases surface O3 levels under low-surface O3 conditions due to the downwards transport of tropospheric O3 lying above the mixing layer, while it decreases O3 under high-surface O3 conditions due to the increased vertical dispersion of surface O3. Changes in ventilation have stronger effects on particulate matter (PM) than on O3 because of the lower PM background concentrations (Jacob and Winner, 2009). Increasing ventilation rates can have a diluting effect on PM, as well as a compensating effect that is associated with a decrease in precipitation, leading to reduction in removal via wet deposition. Dry deposition of both gases and PM are also dependent on the surface conditions, where e.g. a decrease in the snow cover can lead to an increase in the deposition over land. Opposite will a decrease in the sea ice extend result in a decreased removal over marine areas as O3 deposit slower to water surfaces than to sea-ice.

PM concentrations decrease with increasing precipitation as wet deposition is the main removal process for PM, in addition to sedimentation that affects coarse particles. Summertime PM levels are highly sensitive to precipitation frequency, when events tend to be convective and small in scale, while they show

a low sensitivity in winter when synoptic-scale storms dominate. Changes in humidity and cloudiness also affect PM concentration and properties. Increasing relative humidity increases the PM water content and thus, the uptake of semi-volatile components, mainly for nitrate and organics. Sulfate can be produced by the aqueous-phase oxidation of SO2 by hydrogen peroxide (H2O2) in the cloud. However, due to the short time scale of this process (i.e. minutes), it is very challenging to simulate aqueous-phase sulfate formation in clouds in mesoscale or global models (Koch et al., 2003).

2. Climate change impacts on ozone

Atmospheric O3 and its sensitivity to climate changes has gained increasing attention throughout the last decades. First in simplified sensitivity studies were one parameter (e.g. temperature) was changed in chemistry transport models (CTM) and then in more complex studies using chemistry-climate models directly or by using CTMs driven by meteorological 3-D fields projected by climate models (e.g. Hedegaard et al. 2008). Some studies have looked at the global scale, while more studies have focused on projections of the chemistry-climate impacts over polluted regions in mainly the U.S. and Europe. In order to isolate the effect of climate change, the anthropogenic emissions have typically been kept constant in these studies. In more recent years a number of reviews have gone through the literature summarizing the latest findings regarding the climate change penalty on surface O3. We will in the following focus on recent findings for Europe and give an overview of the related uncertainties and knowledge gaps.

As shown by Otero et al. (2016), certain types of synoptic and meteorological drivers favor higher concentrations of extreme ozone. In particular atmospheric blocking favors the occurrence of heatwaves and high ozone concentrations in the summer (Pfahl and Wernli, 2012), both key phenomenon for EXHAUSTION. Pfahl and Wernli (2012) quantified that up to 80% of summer warm extremes are collocated with blocking. Climate models have notoriously issues representing atmospheric blocking, but the literature suggests that the frequency of blocking events in Europe will not change dramatically in the future, and neither will the relationship between blocking events and heatwaves (Schaller et al., 2018). However, with global warming, heatwaves will get more intense, more frequent and last longer (Russo et al., 2014), increasing the likelihood for periods with higher ozone concentrations and high temperatures, especially during blocking events.

Colette at al. (2015) analyzed the robustness of the O3 climate penalty for Europe by setting up a meta-analysis of 25 model projections published in 11 scientific papers (from 2007 and onwards). This combined ensemble included both regional and global CTMs driven by a number of different climate projections for several periods towards 2100. The spread of the ensemble was therefore relatively large, but the robustness of a climate penalty on O3 across large parts of the continental Europe was clear. Over continental Europe, a summertime penalty in the surface O3 levels of [0.44 – 0.64] and [0.99 – 1.50] ppbv was found, when given as the 95% confidence interval for the periods 2041-2070 and 2071-2100 and as an average over Europe. However, these numbers include a latitudinal tendency in the penalty. In Southern and Central Europe, the climate penalty is more pronounced and by the middle of the century the summertime change in O3 can exceed + 1ppbv. In contrast, a small decrease in the summertime O3 level was projected at more northern latitudes (Scandinavia and British Isles). Colette and colleagues also concluded that based on the available model results it was not possible to establish a direct and significant relation between the average European temperature and O3, underlining the importance of other meteorological parameters and their impact on O3.

Figure 2 summarizes the recent findings on the impacts of climate on the future levels of European O3, adopted from Fu and Tian (2019). As part of the FP7 funded project IMPACT2C, the potential future developments in air pollution levels in Europe were studied in a multi-model setup including four state-of-the art chemistry-transport models (Watson et al., 2016; Lacressonnière et al., 2016). Assuming +2ºC global warming, the model simulations pointed towards a modest increase in the averaged European O3 levels during summer (- 0.1 to + 0.8 ppbv relative to present day), but the results were not statistically significant across the four models (Watson et al. 2016). When analyzing the SOMO35 (sum of daily maximum O3 exceeding 35 ppmv) in more details, Lacressonnière et al. (2016) described that three of the four models simulate an increase in summertime O3 across Europe and for parts of central Europe, Eastern Europe and in Spain all the models agrees on higher surface O3 levels during the summer. All models indicate smaller changes during winter.

In a more recent study, Fortems-Cheiney et al. (2017) have used the CHIMERE chemistry-transport model in combination with the RCP8.5 and RCP4.5 climate scenarios combined with anthropogenic emission scenarios in order to investigate the impact of a + 3ºC and +2ºC temperature change. Based on simulations including emissions for 2050 (the ECLIPSE-v4a CLE) the differences in the annual mean European O3 levels between a +2ºC (31.99 +/- 1.01 ppbv) and +3ºC (34.41 +/- 1.07 ppbv) scenario are statistically significant. On average the levels are 8% higher in the +3ºC scenarios, illustrating the impact of the higher temperature alone (the impact from emissions will be discussed below). In relation to health issues, it is important to note that this study also finds that the frequency of days with high O3 levels are increasing in the +3ºC scenario leading to e.g. a higher number of days where the WHO limit value for ozone is exceeded in especially the South-eastern part of Europe. Fortems-Cheiney et al. (2019) concluded this was linked to a lower boundary layer height and a larger number of very hot summer days in this region in the warmer scenario.

In the recent review by Fu and Tian (2019) on the current understandings of the climate change penalty to O3, the emission of biogenic VOCs and the driving feedback processes is listed as one of the key knowledge gaps. The terrestrial biosphere emits BVOCs as part of photosynthetic activity and the emission is a function of e.g. plant species as well as a number of environmental parameters like temperature and sunlight (Hantson et al., 2017). The natural emissions of biogenic VOCs (e.g. isoprene and monoterpene) are therefore by nature highly variable in space and time. A warming climate can directly influence the emissions of biogenic VOCs and thereby O3 levels (Im et al., 2011). The BVOC emissions is typically calculated on-line in the chemistry-transport models and several of the studies referred to above include a significant increase in the BVOC emissions in the projections for the future. However, the sensitivity to temperature can differ substantially between chemistry-transport models (Langner et al. 2012). Newer studies use more complex and dynamic vegetation models in order to assess the future developments in biogenic VOC emissions, when taking other drivers than temperature into account (e.g. Bauwens et al., 2018; Hantson et al., 2017). They show that increasing atmospheric CO2 concentrations can lead to increased biogenic VOC emissions (the greening effect), but similar can the inhibitory CO2 effect work against this increase. The overall impact of processes is still not well understood. Likewise, natural and anthropogenic induced vegetation changes can lead to changes in the

distribution and strength of the biogenic VOCs emissions across Europe in the future. These studies indicate that the strong increase in the emissions in some cases might be exaggerated and that the projections for future developments in biogenic VOCs still are very uncertain.

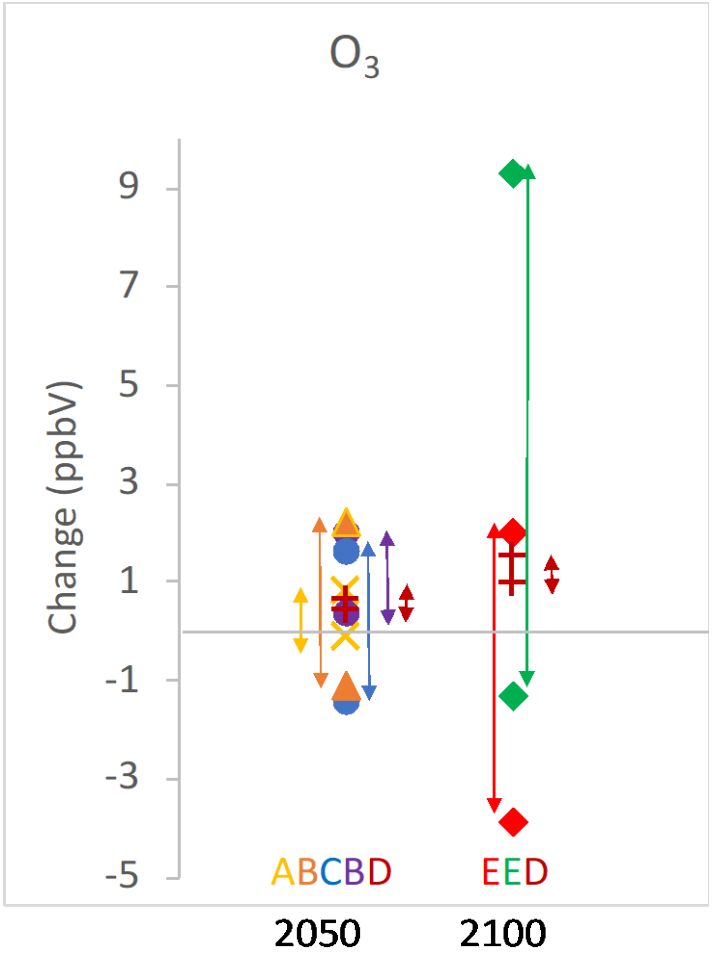

Figure 2. Model projections of O3 as calculated by the studies included in this assessment report, based on Fu and Tian. (2019). The changes are color coded by the studies; A: Watson et al. (2016), B: Fortems-Cheiney et al. (2019), C: Glotfelty et al. (2016), D: Colette et al. (2015), and E: Schnell et al. (2016). Arrows denote ranges reported by the study.

3. Climate change impacts on particulate matter and components

Impacts of climate on aerosol levels have been extensively studied the last few decades. However, different studies focused on different time horizons (e.g. mid-century or end of the century), using different climate scenarios, and different chemistry and transport models (CTMs) that are either coupled to global/regional climate models, or CTMs coupled to global climate projections downscaled to European scale. This section focuses on estimates of the isolated impacts of climate on the particulate matter (PM) levels and their chemical composition over Europe (i.e. model studies where emissions of PM precursors are kept constant). Figure 3 summarizes the changes in PM and its components as calculated in the different studies. In general, studies agree in the direction of the response, regardless which climate scenario has been implemented, with business as usual scenarios giving the largest changes for the anthropogenic species. Dust relatively stays constant throughout the century, while sea-salt is projected

to increase. It can also be seen that most studies calculated a weak impact of climate change on the chemical composition, while the level of confidence varies from low to medium as discussed in Jacob and Winner (2009), Fiore et al. (2012), and Isaksen et al. (2019), as summarized in Table 1.

In the near-future (2030), PM_{10} (PM with a size below 10 μ m) concentrations are projected to decrease

by 0.6 μg m^{-3} (summer) to 4.5 μg m^{-3} (winter) (Lacressonnière et al., 2014), while in 2050, PM_{10} concentrations are projected to decrease by 5.1 μg m^{-3} to 0.1 μg m^{-3} (Juda et al., 2012: Lacressonnière et

al., 2014 and 2017; Cholakian et al., 2019). These differences are attributed to increases in winter precipitation (Juda et al., 2012: Lacressonnière et al., 2014 and 2017), increases in humidity (Lacressonnière et al., 2014), or changes in wind patterns and mineral dust emissions (Lacressonnière et al., 2017). Very recently, Cholakian et al. (2019) projected a decrease of PM_{10} concentrations by 0.01 μg m^{-3}, 0.01 μg m^{-3} and 0.42 μg m^{-3} under the RCP2.6, RCP4.5 and RCP8.5 climate scenarios, respectively. Their analyses showed that these changes can vary seasonally and by the different aerosol species comprising the PM mass. They showed that when the impacts of climate change are isolated, nitrate decrease governs the decrease of PM_{10} and $PM_{2.5}$ in RCP4.5 and RCP8.5; however, in RCP2.6, the increase in dust, sea-salt and biogenic SOA particles outweigh the decrease in nitrates.

$PM_{2.5}$ (PM with a size below 2.5 μm) concentrations are projected to increase slightly in most studies. There are, however, also studies showing decreases, but in smaller magnitudes compared to the increases. Silva et al. (2017) showed that population-weighted $PM_{2.5}$ concentrations will increase by 0.3 μg m^{-3} in 2030 under the RCP8.5 scenario. By the mid-century, $PM_{2.5}$ levels are projected to change by -1.6 μg m^{-3} to 2.47 μg m^{-3} (Lacressonnière et al., 2016; Lemarie et al., 2016; Silva et al., 2017). Lacressonnière et al. (2016), using three regional CTMs coupled with climate models, calculated a change in $PM_{2.5}$ concentrations of -1.6 μg m^{-3} to 2.47 μg m^{-3} under RCP4.5 scenario, with ensemble mean of -0.5 μg m^{-3} to 1.3 μg m^{-3}. They have found an increase of $PM_{2.5}$ over south Western Europe, while a decrease is projected over some parts of eastern and central Europe. They showed that the largest differences due to changes only in climate are due to dust and sea salt, which are largely dependent on meteorological fields.

Lemarie et al. (2016) projected a decrease of area-weighted $PM_{2.5}$ concentrations by 0.98 μg m^{-3} by the end of the century, mainly attributed to changes in surface temperature. They suggested the strong influence of high vertical stability events, where low surface temperature and planetary boundary layer (PBL) height leads to an increase in $PM_{2.5}$ concentrations, while for high temperature ranges, the PBL height becomes a less discriminating factor. However, they suggest that even though the PBL depth constitutes the most important meteorological driver for $PM_{2.5}$, it does not evolve notably compared to the surface temperature in the future. Recently, Park et al. (2020), using a set of ACCMIP global models, showed that in 2100, $PM_{2.5}$ concentrations over Europe will increase on average by 0.13 μg m^{-3} (5.1%), ranging from -0.23 μg m^{-3} to 0.50 μg m^{-3}, under the RCP8.5 scenario, in agreement with Westervelt et al. (2016) who calculated an increase of 0.19 μg m^{-3} in $PM_{2.5}$ concentrations. Westervelt et al. (2016) also showed a decrease of $PM_{2.5}$ concentrations by 0.2 μg m^{-3} in 2100 if RCP2.6 is used instead of RCP8.5.

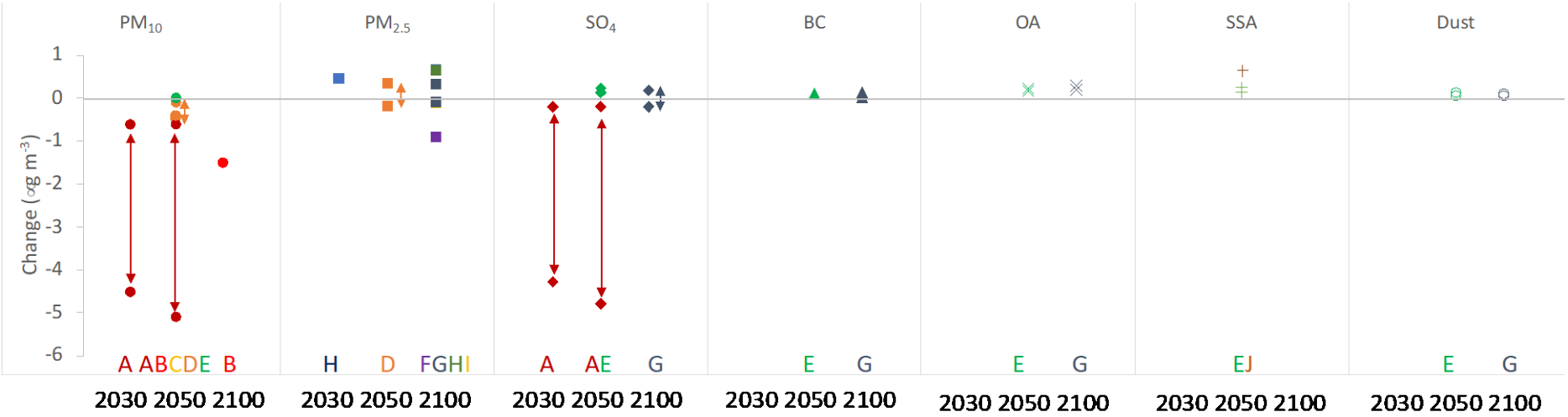

Figure 3. Model projections of PM and its components as calculated by the studies included in this assessment report. The changes are color coded by the studies; A: Lacressonniere et al. (2014), B: Juda et al. (2012), C: Manders et al. (2012), D: Lacressonniere et al. (2017), E: Cholakian et al. (2019), F: Lemaire et al. (2016), G: Westerveld et al. (2016), H: Silva et al. (2017), I: Park et al. (2020), J: Soares et al. (2016). Arrows denote ranges if reported by the study.

Megaritis et al. (2014) investigated the impact of different meteorological variables on $PM_{2.5}$ concentrations, perturbing temperature, humidity, precipitation and wind speeds individually, in prescribed increments. They showed that in Europe, $PM_{2.5}$ concentrations change by -0.7 to 0.3 μg m^{-3} per 1 Kelvin increase in temperatures, while they change by -0.13 to 0.16 μg m^{-3} per 1% change in humidity, by -0.11 mg m^{-3} per 1% increase of precipitation and by 0.14 μg m^{-3} per 1% increase of wind speed. They concluded that precipitation is expected to have the largest impact on $PM_{2.5}$ concentrations under a changed future climate, driven mainly by the accelerated wet deposition of $PM_{2.5}$ species and their gas precursors. They showed that $PM_{2.5}$ concentrations can change by up to ±2 μg m^{-3} in the future due to changes in precipitation. In addition, they projected that changes in wind speed over Europe in the future can change $PM_{2.5}$ levels up to ±1.4 μg m^{-3}. Similarly, Im et al. (2012) calculated that higher temperatures increase biogenic emissions, thereby enhancing biogenic SOA by 0.01 ± 0.00 μg m^{-3} K^{-1} and NO_3^- aerosol concentrations by 0.02 ± 0.02 μg m^{-3} K^{-1}. They also showed that increased temperatures reduce nss-SO_4^{2-} by 0.04 ± 0.07 μg m^{-3} K^{-1}, induced by significant reduction in the cloud cover (90% K^{-1}) and subsequent aqueous-phase production. The $PM_{2.5}$ concentrations show a very small positive response to temperature changes, increasing by 0.003 ± 0.042 μg m^{-3} K^{-1} (0.04% K^{-1}) due to the compensation of organic carbon increases by nss-SO_4^{2-} reductions. They also showed that locally, larger changes can take place, with nss-SO_4^{2-} and NO_3^- in fine aerosols reduced by up to 0.62 μg m^{-3} K^{-1} and 0.80 μg m^{-3} K^{-1}, respectively. Increases as high as 0.097 μg m^{-3} K^{-1} and 0.034 μg m^{-3} K^{-1} are calculated for organic and elemental carbon, respectively. Results show that changes in temperature modify not only the aerosol mass but also its chemical composition.

Regarding SO_4^{2-} aerosols, studies are not consistent in the direction of the change in concentrations in response to changes in climate. Lacressonniere et al. (2014) projected a decrease of SO_4^{2-} concentrations by 4.2 to 0.3 μg m^{-3} in 2030, while in 2050, they projected a slightly larger decrease, which they attributed to higher precipitation and humidity in 2050 compared to 2030, enhancing the transformation of SO_2 into SO_4^{2-} aerosols. Cholakian et al. (2019), on the other hand, projected an increase of SO_4^{2-} by 0.02 μg m^{-3} (RCP8.5) to 0.11 μg m^{-3} (RCP4.5) in 2050, based on the different RCP scenarios they have used, attributed to higher humidity in winter and OH levels in summer due to increased temperatures. They also showed that decreased mixing heights contributed to these increases.

All the different studies agreed on a likely increase of organic aerosols (OA), due to increased temperatures, and thus increase in biogenic emissions and OH levels. While increased temperatures lead to increased biogenic emissions, which in return consume the available OH, it also leads to increased OH by regeneration chemistry by increasing O_3 in areas with sufficient levels of NOx. Cholakian et al. (2019) projected an increase of 0.06 μg m^{-3} to 0.12 μg m^{-3} in 2050 under the different RCP scenarios. Westervelt et al. (2016) projected end of the century increases of 0.09 μg m^{-3} to 0.19 μg m^{-3} under RCP2.6 and RCP8.5 scenarios, respectively. Regarding sea-salt concentrations, these slightly increase by 0.01 to 0.11 μg m^{-3}, due to increases in wind speeds (Cholakian et al., 2019) and sea-surface temperatures (Soares et al., 2016), leading to increased sea-salt emissions. Regarding dust concentrations, Westervelt et al. (2016) shows a slight increase in 2100 according to the RCP2.6 scenario due to an increase in wind speeds, while in RCP8.5, a decrease is projected due to a decrease in wind speeds.

Table 1. Impacts of temperature-driven pathways on surface O_3 and PM, adapted from Jacob and Winner (2009), Isaksen et al. (2009) and Fiore et al. (2012). Symbols depict consistently positive (++), generally positive (+), weak or variable (=), generally negative (-), and consistently negative (--).

	Ozone	PM	Wildfires	Level of Confidence
Temperature	++	-	++	Medium
Regional Stagnation	++	++		Medium
Wind Speed	-	-		Low
Mixing Height	=	- -		Low
Humidity	=	+	- -	High
Cloud Cover	-	-		Low
Precipitation	=	- -	- -	Low
BVOC Emissions	+	+		Low
Wildfires	++	+		Medium

4. Impacts on air pollution due to changes in emissions vs changes in climate

Several of the model studies included in this review have also analyzed the combined effect of future changes in both climate and emissions. For Europe, the anthropogenic emissions of the main air quality components (e.g. SOx, NOx and PM2.5) are projected to decrease in the future posing the central question whether the benefits of these emission reductions are counteracted by a possible climate penalty.

Cholakian et al. (2019) analyzed the effect of both a current legislation and maximum feasible reduction emissions scenario combined with climate scenarios and impacts of long-range transport from outside Europe. They concluded that the anthropogenic emissions have the largest impact on the European PM concentrations towards the end of the century, compared to developments in both climate and long-range transport. In the global study by Westervelt et al. (2016) a number of simulations with a coupled chemistry-climate model for different climate and emissions scenarios leads to a similar conclusion for PM2.5: the climate impacts on PM2.5 are relatively minor compared to emissions impacts.

In the Colette et al. (2015) review of previous studies including the combined effect of climate and anthropogenic emissions developments, they concluded that expected emission changes will have the largest impact on European O3 levels. This conclusion is, however, very dependent on the scenarios used for Europe, but also for the developments outside Europe. In the study by Fortems-Cheiney et al. (2017), the high emission scenario RCP8.5 and the related higher European O3 concentrations, would to a large degree be caused by inflow from outside Europe. In the more detailed study by Lacressonniere et al. (2014), it is also shown that long-range transport from outside Europe can increase the O3 levels in Northern Europe in spite of decreasing European emissions. The impact of the European emissions was studied directly by Fortems-Cheiney et al. (2017) as they were running a CTM with both current day (2005) and future (2050) European emissions and future climate data. The applied Current Legislation emissions for 2050 projects a significant decrease (more than 40%) in most components across Europe (except for the Turkish area). When emissions are fixed to 2005 under RCP8.5 for 2050, the mean summer O3 levels increase by up to + 12% (+4.30 ppbv), except in urban areas with high NOx emissions.

The future developments in health impacts related to air pollution have also been assessed in a number of studies (see e.g. the review by Doherty et al., 2017 and the global study by Silva et al. 2017). Geels et al.

(2015) combined two chemistry-transport model with a health assessment model and focused on the development in the number of premature deaths related to PM2.5 and O3 in Europe in the 2050s and 2080s. Climate change alone in the RCP4.5 scenario, where emissions were kept constant, had only minor impacts (+/- 2-4%) on PM2.5 related number of cases, while the number of cases related to O3 increased by up to 10% (relative to the 2000s). Including also the emissions from the same RCP4.5 scenario leads, however, to significant decreases in the health effects especially for PM2.5 with a significant decline in number of premature deaths (~65% in 2050s and ~80% in 2080s). But, as pointed out by Doherty et al. (2017), the health assessments for future conditions are related to additional uncertainties as e.g. the temperature effects on applied pollutant-response relationships are not well known.

5. Feedbacks from air pollution to climate

Air pollutants like O3 and different types of aerosols (e.g. black carbon (BC), SO42-, NO3- and organic aerosols (OA)) have important radiative effects on climate, while other pollutants such as carbon monoxide (CO), VOCs, nitrogen oxides (NOx) and SO2, although are negligible greenhouse compounds themselves, have an important indirect effect on climate by altering the abundances of radiatively active gases such as O3 and CH4, a well as by acting as precursors for secondary inorganic and organic particles (Stohl et al., 2015).

Tropospheric climate–chemistry interactions are to a large extent related to chemical and physical processes and to compounds that show large variability in global distribution and trends, mainly driven by precursor emissions that also show large spatial variability. Atmospheric composition influences climate by regulating the radiation budget and cloud properties (Boucher et al., 2013; Stohl et al., 2015), through the presence of water vapor (H2O), CO2, CH4, O3, and aerosols. The main radiative effect of the gases is through the greenhouse effect (i.e. through interference with long-wave radiation), thus heating the atmosphere, while aerosols may either heat or cool the surface (through interference with short-wave radiation), depending on their optical properties (Table 2: Bond et al., 2013; Stohl et al., 2015). Aerosol–cloud interactions represent an area with potential for strong interactions in the climate system (Myhre et al., 2013). Aerosols determine the cloud radiative properties and thus the radiative heating/cooling, and participate in the precipitation process. Although considerable progress has been made in recent years to include parameterization of aerosol–cloud droplet interaction and explicit microphysics for cloud water/ice content in climate models, inadequate understanding of the processes contributes to significant uncertainties in model simulated future climate changes (Table 2: Malavelle et al., 2017; Toll et al., 2019).

Regarding the trends in emissions of air pollutants affecting radiative forcing, large differences are observed in the temporal and geographical distributions of human-related (anthropogenic) emissions and natural emissions (Shindell and Faluvegi, 2009; Flanner, 2013). Anthropogenic emissions respond to regulations and to different growth rates in energy use, transportation, and industry. For example, emissions from Europe generally show reductions thanks to the regulatory actions taken over the decades (EEA, 2019b), while emissions in regions like Southeast Asia, and in other developing countries show large increases during recent years (IEA, 2020). In addition to geographical differences, there are also sectoral

differences, such as the large increases observed in aircraft and shipping (Isaksen et al., 2009). On the other hand, natural emissions of key climate precursors (NOx, CO, biogenic VOCs and sulfur compounds) are difficult to regulate and exhibit a large year-to-year variations (e.g. in the case of biomass burning).

6. Climate change impacts on wildland fires

Climate warming causes an increasing number of wildfires in some regions (Lozano et al., 2017) because it increases wildfire risk by raising the average temperature of the atmosphere, intensifying droughts, and prolonging the dry season. Driving mechanisms behind the fires varied in different periods. Pechony et al. (2010) argued that in the preindustrial period fires were strongly driven by lightning, rather than temperature. With the Industrial Revolution, this changed to an anthropogenic-driven regime, and there are now indications that we are moving towards a temperature-driven fire regime during this century. However, currently the main cause of wildfires is anthropogenic and result from agriculture and forest management practices, as well as from recreational habits (Sofiev, 2013).

Modelling the impacts of climate change on wildland fires is very challenging due to the complex dynamics of fuel accumulation, vegetation dynamics and their interactions with changing climate, as well as the difficulty in representing the fire behavior patterns due to impacts of land management and human ignitions (Hantson et al., 2020; Sanderson and Fisher, 2020). Thus, projection of wildland fires over decadal to century timescales requires more mechanistic approaches, capable of capturing the numerous interacting system components that affect the evolution of fire risk. Process-based global fire models based on these principles have progressed rapidly over the past decade, although their use in fully coupled earth system models is still not standard practice (Sanderson et al., 2020), leading to omitting a potentially important component of the global carbon–climate feedback.

Table 2 Impacts of gases and aerosols on the top of the atmosphere (TOA) radiative forcing, adapted from Myhre et al. (2013). Symbols depict consistently positive (++), generally positive (+), weak or variable (=), generally negative (-), and consistently negative (--), following Jacobs and Winner (2009).

Pollutants	TOA Radiative Forcing	Level of Confidence
CO_2	++	Very high
CH_4	++	High
CO	++	Medium
NMVOC	++	Medium
NOx	=	Medium
O_3	=	Medium
SO_4^{2-}	- -	High
BC	++	High
OA	--	High
Mineral Dust	=	High
Aerosol-cloud interactions	- -	Low

Interaction between the anthropogenic activity, landcover, meteorological conditions and intensity of wild fires (expressed as the amount of biomass burned or pollutants released) basically resides to the human activities playing the fire ignition and suppression roles whereas vegetation and weather modulate

the natural fire lifetime and potential for spreading (Balch et al., 2017; Fusco et al., 2016; Prestemon et al., 2013). Most studies that have estimated future fires (risk or area burned) have relied on statistical regressions. However, it is highly regionalized. For instance, in Indonesia (one of the largest sources of fire-induced pollution worldwide) the use of fire for clear and prepare land on degraded peat is a routine

repeated every year – with fires routinely getting out of control. However, it is the El-Nino-induced drought that can make them devastating (Field et al, 2015). In the Western US, worsening climatic conditions will lead the fires to become the main source of PM2.5 if the human behavior and practices remain the same (Liu et al., 2016; Yue et al., 2013). Similarly, Ford et al (2018) found that, despite the reduction of anthropogenic PM2.5, raising fire contribution will lead to a net increase of PM2.5 in several US regions.

Wildfires are known to be the largest contributor to global biomass burning. They are large source of atmospheric trace gases and aerosols, in addition to major vegetation disturbance (Knorr et al., 2013). They emit into the atmosphere primarily CO2 among other greenhouse gases and carbonaceous particulate matter among other aerosols (Saarikoski et al., 2007). However, our knowledge of wildfires as an integrated part of the Earth system is still rather weak (Bowman et al., 2009; Langmann et al., 2009; Keywood et al., 2013).

In Europe, climate change is expected to increase the probability for forest fires in areas where increase in summertime temperature is accompanied by decrease in precipitation, as in the Mediterranean area (Mouillot et al. 2002; Bedia et al. 2014, Kalabokidis et al, 2014). In more northern regions the estimations of precipitation amounts are less conclusive (Jylhä et al. 2009; Ylhäisi et al. 2010), which makes predicting the fire sensitivity much more difficult. A global modelling study of Knorr et al (2017), unlike the research quoted above, gives calmer picture. The authors conclude that at the global scale anthropogenic emission reduction will have the more significant effect and the WHO target of 10 µg PM m-3 is in reach in many regions. However, the authors also admitted high uncertainty connected with the fire abatement measures, which they accounted for.

Air pollution due to PM2.5 released from vegetation fires is a notable risk factor for public health everywhere in the world, also in Europe (Johnson et al., 2020). This should be taken into consideration when evaluating the overall health and socio-economic impacts of the fires. Kollanus et al. (2016) evaluated that in 27 European countries, over thousand premature deaths were attributable to the vegetation-fire originated PM2.5 concentrations in 2005 and 2008. Estimated impacts were highest in southern and eastern Europe, but all countries were affected by fire-originated PM2.5. Saarnio et al. (2010) analyzed smoke plumes originated from wild-land fires, which were detected in Helsinki, Finland, during a one-month-lasting period in August 2006. The measurements showed that the major growth in PM concentration was caused by submicrometer particles consisting mainly of particulate organic matter. They reported an increase of a factor of 1.5 to 3.5 in PM2.5 concentrations. Saarikoski et al. (2007) calculated that the wildland fires increased the PM2.5 concentrations by a factor of 3.6 to 4.3. Sofiev et al. (2008) considered the influence of wild-land fires on air quality in Europe, using spring and summer seasons of 2006-2007 as prominent examples. High concentrations of nearly all pollutants were detected in Central, Eastern and Northern Europe in late spring of 2006, causing widespread allergic symptoms and other illnesses associated to poor air quality.

The aerosols produced by fires seem to be different from those emitted from e.g. anthropogenic

industrial or transport sources: intriguing observations of Dusek et al. (2005) showed that peat-fire smoke has much weaker interaction with clouds than one would expect: less than 50% of smoke particles of suitable size were activated to CCN, whereas this number often approaches 100%. Therefore, the fire aerosols may be not a very efficient shielding agents as other types of particles. The other factor is darkening surface after the fires: albeit small (at 1km scale the albedo change is ~0.002 in sub-Saharan

Africa (Gatebe et al, 2014)) these changes occur over wide areas and constitute noticeable additional radiative forcing.

Generally, there is some agreement that carbonaceous aerosol emissions from anthropogenic sources may decrease in the future (Lamarque et al., 2011; Bond et al., 2013; Shindell et al., 2013). Global total carbonaceous emissions are expected to be reduced due to new, less polluting technologies along with a shift away from the burning of coal and wood in the residential sector, while this reduction will vary regionally and not be reduced everywhere (Streets et al., 2004). However, the future trends of wildfires and their aerosol emissions is more uncertain due to complicated interactions between fire, land, atmosphere, and, most importantly, anthropogenic stress. Even now the state-of-the-art Earth system models have very limited capacity to simulate wildfire - climate interactions (Carslaw et al., 2010).

Estimating the fire emission heights is of particular importance when evaluating the coupling between terrestrial and atmospheric processes. Emissions heights affect the aerosol long-range transport, aerosol-cloud interaction, and radiation (Luderer et al., 2006; Samset et al., 2013). One large problem is that emission heights are strongly affected by fire intensity and atmospheric stability. As illustrated by Joshi et al. (2007) and Luo et al. (2013), both of these are expected to change in the future. The emission height and its effect on atmospheric black carbon and its transport has been evaluated using present day climate conditions by Veira et al. (2015a,b), Sofiev et al. (2012, 2013), Kukkonen et al. (2014), and Daskalakis et al. (2015). Attempts to apply that same approach to future climate has more recently been done by Veira et al. (2016).

The radiative forcing due to aerosols originating from wildfires has mostly been investigated using present-day climatology and prescribed wildfire emission inventories (satellite based). Based on the literature review by Veira et al. (2016), the most recent climate-aerosol models state that the total wildfire top of the atmosphere radiative forcing range between +0.18 Wm-2 (Tosca et al., 2013) and -0.29 Wm-2 (Jones et al., 2007).

7. Co-benefits and trade-offs related to GHG and air pollution mitigation and adaptation policies

There is a general consensus among studies that there are additional (co) benefits from mitigation of GHG emissions in terms of improved air quality (Bollen 2015; Klausbruckner et al. 2016; de Oliveira and Doll 2016; Rao et al. 2017; West et al. 2013).The estimated improvements in air quality under different climate mitigation scenarios depend on socioeconomic and technological development and the assumptions on the development of air pollution in the next few decades. Rao et al. (2017) examined the impacts of different air pollution narratives under the global Shared Socio-economic Pathway (SSP) framework, concluding that scenarios with limited climate change mitigation present higher air pollution impacts than scenarios with more stringent mitigation trajectories. An overview paper (von Schneidemesser et al., 2015) concludes that any policy actions intended to mitigate one of these two issues must necessarily take into account the feedbacks with the other, to avoid that benefits to one sector, will worsen the situation in another.

The effect of air pollution policies on climate have been investigated in some studies. Shindell et al. (2012) identified 14 emission control measures targeting short- lived climate pollutants (SLCPs) and relative to their reference scenario, found that all CH4 and BC control measures avoid a global mean surface temperature increase of ~0.5°C by 2050, although significant uncertainties exist regarding the extent to which reductions of SLCFs impact climate change (Myhre et al., 2013). In light of the estimated large adverse

health impacts of air pollution around the globe, emission controls on SO2, BC, and O3 precursors may continue to be imposed irrespective of climate policy (Anenberg et al., 2010; Silva et al., 2013).

Reduction of non-SLCF pollutants, such as NO2 and SO2, generally leads to exacerbation of climate change because these species are the key aerosol precursors thus responsible for almost 1 W m-2 or shielding from the solar radiation roughly-equally split between direct and indirect effects (IPCC AR5, Physical science basis). The secondary inorganic aerosols mainly consisting of nitrated and sulphates are highly reflective and excellently soluble, thus facilitating both direct (reflection of sunlight back to space) and indirect (increasing the cloud albedo and modifying the cloud cover and rain patterns) aerosol effects. Recent studies showed that, e.g., The International Convention for the Prevention of Pollution from Ships – Annex VI (MARPOL-VI) required reduction of sulfur content in ship fuels brings substantial health benefits (global avoided mortality 74-200 thousand cases annually, 95% confidence interval) but adds about 0.07 W m-2 due to lost cooling effects of sulphates (Sofiev et al, 2017). Estimates of other studies of the shipping cooling effect showed similar results suggesting the range from ~0.04 W m-2 up to 0.11 W m-2 (Lund et al, 2012, Fuglestvedt et al, 2008, Eyring et al, 2007, Skeie et al, 2009).

Several studies show potential air quality co-benefits in Europe, including in the UK and Ireland (Karlsson et al., 2020; Alam et al., 2017, 2018; Wilkinson et al., 2009) and the Nordic countries (Åström et al., 2013; Williams, 2007). Rafaj et al. (2014) associate large potential co-benefits in Europe with renewable energy, due to reduced mercury emissions to air. Collette et al. (2013) looked at combined projections of air quality and climate impact at the regional scale over Europe under the CMIP5 climate scenarios and conclude that exposure to air pollution will decrease substantively by 2050 according to a mitigation pathway that aims at keeping global warming below 2◦C by the end of the century where exposure weighted SOMO35 and PM2.5 are reduced by 80 and 78%, respectively in Europe. Large potentials for reaping air quality co-benefits are also shown for several European cities, e.g. Rotterdam (Tobollik et al., 2016) and Barcelona, Malmö, Sofia and Freiburg (Creutzig et al., 2012). Markandya et al. (2018) estimated that the co-benefits would make a valuable contribution towards covering the mitigation costs, from 7% to 84% in the EU-27 countries.

The impacts of climate adaptation policies on greenhouse gas emissions and air pollution has also been discussed. Classic adaptation strategies to increased temperatures like air conditioning have been shown to result in increased GHG emissions as well as increased air pollution (Abel et al., 2018, Watts et al., 2019). In a study in Eastern USA, Abel et al. (2018) found that in a warmer climate 3.8% of the total increase in PM2.5 and 6.7% of the total increase in O3 are attributable to extra air conditioning use. Other climate adaptation actions like increasing social capital to reduce health risks from natural hazards and improving urban design to reduce heat stress can have significant co-benefits for air pollution (Cheng and Berry, 2013).

8. Health-related advance warning systems in case of extreme weather conditions

Advance warning systems in case of extreme meteorological conditions can be used in preparing for and mitigating the health impacts to the public. As the occurrence of various extreme meteorological conditions will increase with the climate change, the development and use of such systems has to be an important part of the adaptation and mitigation strategies and policies. This section presents a review of such systems that are currently in use in Europe.

8.1. Role of national weather services

National Meteorological and Hydrological services (NMHS) have typically legal status as an authority and obligations to provide (hydro)meteorological observations, forecasts and warning in their country in order to promote safety of the citizens. However, the role of NMHS and set of their services vary from country to country and in many countries, separate institutions provide meteorological and hydrological services. Very few NMHS have expertise in air quality and are able to provide air quality forecasts and warnings.

Weather observations, forecasts and related warnings are obvious core functions of the national weather services. Traditionally NMHSs have concentrated on predicting only weather hazards, but due to growing societal demand, there is ongoing shift of paradigm towards developing impact-based forecasting and impact warnings (WMO-No. 1150, 2015). When thinking about research themes of EXHAUSTION and future possibilities to develop and provide operational, combined heatwave–air quality forecasts and warnings for the health sector, it is obvious that collaboration between institutions and disciplines is required.

8.2. Severe weather warnings for Europe

Meteoalarm is a platform where nationally issued weather-related warnings are collected and presented in coordinated manner. It was developed by EUMETNET a collaboration of 24 national weather services in Europe. Figure 4 shows an example of an issued Meteoalarm warning from EUMETNET. Meteoalarm includes a range of weather hazards from wind, snow and thunderstorms to heavy precipitation and floods and avalanches. Current health-related hazards and warnings, relevant for EXHAUSTION themes as well, include hot weather warnings, cold weather warnings, and forest, grass or wildfire warnings (Table 2).

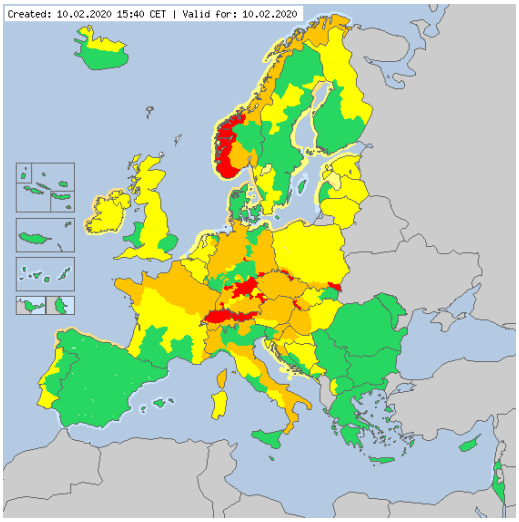

Figure 4 Meteoalarm warnings issued on 10 Feb, 2020. (https://www.meteoalarm.eu/). Countries with grey colors do not contribute to Meteoalarm. (Green: No particular awareness of the weather is required; Yellow: The weather is potentially dangerous; Orange: The weather is dangerous. Unusual meteorological phenomena have been forecast. Damage and casualties are likely to happen; Red: The weather is very dangerous. Exceptionally intense meteorological phenomena have been forecast. Major damage and accidents are likely.)

Table 3 Temperature (hot and cold weather) and forest, grass or wildfire warnings produced by European National Meteorological and Hydrological Services. Green-filled boxes indicate countries that issue a warning, pink-filled boxes indicate countries that do not issue warning, and stripped boxes indicate countries where the responsibility of issuing a warning is not at the NMHS. (Source: Kaija, H., 2020, FMI)

Hot weather warnings produced by National Weather Service							
Austria	Belgium	Bosnia-Herzegovina	Bulgaria	Croatia	Cyprus	Czech republic	Denmark
Estonia	Finland	France	FYROM	Germany	Greece	Hungary	Iceland
Ireland	Israel	Italy	Latvia	Lithuania	Luxemburg	Malta	Moldova
Montenegro	Netherlands	Norway	Poland	Portugal	Romania	Serbia	Slovakia
Slovenia	Spain	Sweden	Switzerland	UK			
Cold weather warnings produced by National Weather Service							
Austria	Belgium	Bosnia-Herzegovina	Bulgaria	Croatia	Cyprus	Czech republic	Denmark
Estonia	Finland	France	FYROM	Germany	Greece	Hungary	Iceland
Ireland	Israel	Italy	Latvia	Lithuania	Luxemburg	Malta	Moldova
Montenegro	Netherlands	Norway	Poland	Portugal	Romania	Serbia	Slovakia
Slovenia	Spain	Sweden	Switzerland	UK			
Forest, grass or wildfire warnings produced by National Weather Service							
Austria	Belgium	Bosnia-Herzegovina	Bulgaria	Croatia	Cyprus	Czech republic	Denmark
Estonia	Finland	France	FYROM	Germany	Greece	Hungary	Iceland
Ireland	Israel	Italy	Latvia	Lithuania	Luxemburg	Malta	Moldova
Montenegro	Netherlands	Norway	Poland	Portugal	Romania	Serbia	Slovakia
Slovenia	Spain	Sweden	Switzerland	UK			

9. Outlook and Recommendations

As this review outlines, the projections of how climate and health relevant air pollution components (e.g. PM2.5 and O3) will develop in the future across Europe, are still associated with large uncertainties. The differences in the modelled projections can to a large degree be attributed to the differences in the experimental designs in these studies e.g. in terms of emission and climate scenarios applied. However, the chemistry-transport models also show differences in climate-sensitivity and so far, these differences

have not been analyzed in detail. More advanced ecosystem models have in recent studies showed that the complex feedbacks between ecosystem and changes in climate, CO2 levels, land-use etc. will modify the natural emissions of O3 precursors to a larger degree than what is included in previous model studies. A robust quantification of the direction of this change is not possible with the current understanding and is according to the detailed review by Fu and Tian (2019) one of the main knowledge-gaps related to projections of future surface O3. Generally, more research is needed for further development of climate and air pollution models, in terms of their capabilities in representing different chemical/physical processes and the response to climate change. In order to get more robust projections with current generation models, multi-model studies are needed, where scenarios and projection timelines are harmonized, as well as large ensemble studies, where model systems and scenarios are kept constant. For health impact studies, it is also important to increase the resolution of the driving climate data in order to perform high-resolution simulations of the developments of the main air pollutant in the future. Higher accuracy in the air pollution simulations will also lead to more robust forecasting on extreme weather and natural hazards such as wildland fires. In EXHAUSTION, improved climate models from the ongoing Climate Model Intercomparison Project Phase 6 (CMIP6), along with state-of-the-art regional air pollution models will be used in order to

increase the robustness of the projections of future climate change and air pollution over Europe, which will be further used to calculate the combined impacts of extreme weather and air pollution.

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