https://doi.org/10.5194/egusphere-2024-755
Preprint. Discussion started: 21 March 2024

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The Co-benefits of a Low-Carbon Future on Air Quality in Europe

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Abstract. There is considerable academic interest in the potential for air quality improvement as a co-benefit of climate change mitigation. Few studies use regional air quality models for simulating future co-benefits, but many use global chemistry-climate model output. Using regional atmospheric chemistry could provide a better representation of air quality changes than global chemistry-climate models, especially by improving the representation of elevated urban concentrations. We use a detailed regional atmospheric chemistry model (WRF-Chem v 4.2) to model European air quality in 2050 compared to 2014 following three climate change mitigation scenarios. We represent different climate futures by using air pollutant emissions and chemical boundary conditions (from CESM2-WACCM output) for three Shared Socioeconomic Pathways (SSP1-2.6, SSP2-4.5, SSP3-7.0; a high, medium and low mitigation pathway).

We find that in 2050, following SSP1-2.6, mean population-weighted $PM_{2.5}$ concentrations across European countries reduces by 52% compared to 2014. Whilst under SSP2-4.5, this average reduction is 34%. The smallest average reduction was 18% by following SSP3-7.0. Maximum 6-monthly-mean daily-maximum 8 h (6mDM8h) ozone (O_3) is reduced across Europe by 15% following SSP1-2.6, and 3% following SSP2-4.5, but increases by 13% following SSP3-7.0. This demonstrates clear co-benefits of climate mitigation. The additional resolution allows us to analyse regional differences and identify key sectors. We find that mitigation of agricultural emissions will be key for attaining meaningful co-benefits of mitigation policies, evidenced by the importance of changes in NO_3 aerosol mass to determining future $PM_{2.5}$ air quality and changes in CH_4 emissions to future O_3 air quality.

1 Introduction

Poor air quality is a major public health issue worldwide. The negative health impacts are usually attributed to two air pollutants - $PM_{2.5}$ (any airbourne non-gaseous particle under 2.5 microns in diameter) which can be both a primary or secondary

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air pollutant and ozone (O_3) which is a secondary pollutant. Primary sources of $PM_{2.5}$ include a range of natural and anthropogenic sources, including fossil fuel combustion, and transport. As a secondary pollutant, it can be formed from emissions of species such as Ammonia (NH_3), Sulfur dioxide (SO_2) and Nitrogen oxides (NO_x). Tropospheric O_3 is a secondary pollutant formed by photochemical reactions involving volatile organic compounds (VOCs) including methane (CH_4), nitrogen oxides (NOx) and carbon monoxide (CO) in the presence of sunlight. Air pollution may have consequences wider than just mortality, such as economic cost (Vandyck et al. 2020) and reduced crop yields (Lobell et al. 2022). Both $PM_{2.5}$ and O_3 are intrinsically linked with climate change; many of the sources of primary $PM_{2.5}$ and O_3 precursors come from the same sources as long-lived greenhouse gases (or in the case of CH_4 are greenhouse gases). Additionally, air pollutants themselves have an impact on climate forcing through a range of pathways including directly affecting the radiative balance of the atmosphere, modifying the albedo of clouds and glaciers and increasing cloud lifetime. (Von Schneidemesser et al. 2020).

Air quality is a major issue due to its impacts on human health, by increasing the risk of a range of diseases. Exposure to air pollution contributes to about 6.7 million deaths per year (World Health Organisation, 2023), 4.2 million of which are from ambient outdoor air pollution and the remainder from household air pollution. In Europe, $PM_{2.5}$ is responsible for an estimated 368,000 deaths per year (Juginovic et al. 2021) and the annual mean mortality rate from air pollution in Europe of 133 per 100,000 exceeds the global mean of 120 per 100,000 (Lelieveld et al. 2020). Additionally, the European Environment Agency (2023) reported that in 2022, 96% of Europe's urban population was exposed to $PM_{2.5}$ concentrations above the World Health Organisation's guideline value of 5 $\mu g/m^3$. (World Health Organisation, 2021). Poor air quality in Europe is therefore still a large factor in human health.

Improving air quality in Europe is feasible: air pollution responds quickly to air pollutant emissions reductions, potentially resulting in lower population exposure to primary air pollutants. Due to reductions in anthropogenic emissions of air pollutants, $PM_{2.5}$ air quality in Europe has improved over the past half-century; between 1960 and 2009, population-weighted $PM_{2.5}$ concentrations in the European Union decreased by 55.3% (Butt et al. 2017). Similarly, the responsiveness of air pollution to emissions changes was demonstrated by the major changes in $PM_{2.5}$ air quality from national to global scales during the COVID-19 pandemic (Jephcote et al. 2021; Venter et al. 2021; Saha et al. 2022). The speed of this response to changes in emissions caused by COVID-19 restrictions indicates that considerable improvements in air quality can be achieved when air pollutant emissions are reduced. Conversely, O_3 concentrations in Europe have increased in the latter half of the 20th century and early 21st century (Turnock et al. 2020) despite considerable reductions in local, anthropogenic O_3 precursor emissions. This is potentially due to increased intercontinental transport of O_3 precursors (Guerreiro et al. 2014). This demonstrates that despite improving trends in $PM_{2.5}$, a different approach may be required to reduce exposure to O_3.

Air pollutants and greenhouse gases have a lot of common sources and there are many linkages between air quality and climate change. It is therefore expected that greenhouse gas mitigation policies may also result in lower air pollution emissions which may improve air quality (Turnock et al. 2020; Vandyck et al. 2020). This co-benefit is often suggested as a motivator

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to encourage faster and more effective climate mitigation action from policymakers, including at the regional level, because it turns the concept of climate change mitigation from a diffuse, global-scale requirement to something that can provide measurable, near-term benefits in the local area (von Schneidemesser et al. 2020). Existing research supports the hypothesis that air quality co-benefits of climate mitigation will be present in Europe. For example, Turnock et al. (2020) finds that across Europe, $PM_{2.5}$ concentrations would decrease by the middle of this century for a range of future scenarios and be stronger in scenarios with greater mitigation. Reddington et al. (2023) finds that reductions in $PM_{2.5}$ across Europe following a sustainable scenario (SSP1-1.9) could improve health across the continent. Fenech et al. (2021) come to a similar conclusion, but focusing particularly on the UK. This may not be true for all pollutants and all scenarios. For example, findings differ for O_3, for which Turnock et al. (2020) project an increase in Europe following scenarios with limited climate change mitigation and Fenech et al. (2021) project an increase compared to the present in all scenarios.

Although improved air quality as a co-benefit of climate mitigation is widely expected, the reality is that the interactions between these policies are complicated and potentially non-linear. Surface level O_3, for example, may worsen following reductions of NOx, which is a common emission from combustion processes, depending on the balance between VOCs and NOx in the vicinity (Miyazaki et al. 2021). Complexity is also added by the interactions between air pollutants and the climate system itself; the climate affects the formation of some secondary air pollutants, including surface level O_3 (Archibald et al. 2020) and secondary organic aerosol (Scott et al. 2018; Raes et al. 2010). Climate change will also affect the prevailing meteorological conditions, and impact the dispersion of air pollutants, thereby affecting human exposure (Graham et al. 2020). Aerosols also affect the atmosphere's radiative balance, causing a cooling that masks the true potential scale of climate change (Peace et al. 2020). It is also not a given that all climate mitigation strategies will reduce emissions of primary air pollutants as it depends on the mitigation strategy used. For example, greater adoption of biofuel burning in the energy sector may not be as effective in improving air quality compared to non-combustion power sources (Buonocore et al. 2021). modelling studies are therefore needed to understand how climate change mitigation and air quality might interact, considering differing strategies for climate mitigation.

There are some key challenges associated with modelling climate co-benefits. One challenge is selecting the assumptions to make about societal development and climate mitigation policies. Using consistent scenarios with the same underlying assumptions makes this easier by improving comparability between studies. Previous research into the linkages between climate change and air quality largely uses the air pollutant emissions associated with CMIP5 (e.g. Silva et al. 2016; Kumar et al. 2018; Fenech et al. 2021), which are linked with the Representative Concentration Pathways (RCPs) (van Vuuren et al. 2011). RCPs are pathways of greenhouse gas concentrations over the 21st Century that result in different radiative forcing endpoints in 2100. Some more recent research (e.g. Rao et al. 2017; Turnock et al. 2020; Reddington et al. 2023) uses the CMIP6 (the successor to CMIP5) emissions, designed to work with the Shared Socioeconomic Pathways (SSPs) (O'Neill et al. 2017). SSPs expand the range of pathways and also provide different narratives of socioeconomic development. These are integrated to provide a matrix of scenarios that each narrative can result in following multiple concentration pathways. This means the

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emissions meant to represent these integrated scenarios factor in the role of socioeconomic development in more detail than in the scenarios used in CMIP5. They also further expand on the link between pollution and climate; providing descriptions on how air pollution control progresses following the narratives, which is then also fed into the air pollutant emissions used in CMIP6 (Rao et al. 2017). The RCPs and CMIP5 emissions did not capture the full set of linkages between climate change and air quality policies due to a lack of detail on socioeconomic development and pollution control and so the SSPs used in CMIP6 will provide a better assessment (Coelho et al. 2023).

Another challenge is the difficulty of modelling air quality and climate simultaneously. Many studies that use the SSPs often use the output from global climate models and/or Earth system models (e.g. Turnock et al. 2020, Allen et al. 2020), or "reduced form" models that generalise over large regions (Rao et al. 2017). These types of models may have less detailed chemistry schemes than models typically used to simulate regional air quality. They also tend to have a coarser horizontal resolution than regional air quality models, which is important for air quality research to simulate chemical processes that impact on air pollutants at local and urban scales (Adedeji et al. 2020, Fenech et al. 2018, Goto et al. 2016). Despite this, global chemistry-climate models have tended to be used for future projections of air quality due to prohibitive computational requirements for running multi-decadal simulations with regional air quality models. Some studies using CMIP6 output (Turnock et al. 2023; Reddington et al. 2023) are making progress in improving the representation and resolution of present-day air quality by combining CMIP6 output with observational and reanalysis data, however, the approaches taken by these studies still use a coarser grid for future simulations. The finer resolution provided by regional models is especially important if the work aims to estimate health impacts as the improved representation of elevated urban concentrations allows for a more realistic population exposure assessment.

A combined approach to modelling co-benefits that uses both regional air quality and global climate models is needed to utilise the advantages of both techniques. An approach allowing the use of air quality models could be isolating the impacts of solely future emissions changes on air quality. It is well established that in Europe, the impact of emissions changes on future $PM_{2.5}$ air quality is likely to far eclipse the impact of climate change (Colette et al. 2013; Chemel et al. 2014; Doherty et al. 2017). Additionally, in Europe, even O_3 pollution may not be sensitive to changes in climate (Zanis et al. 2022), further suggesting that not considering climate change is an appropriate trade-off for improving model resolution and chemistry scheme.

Due to the computational expense of using high-resolution atmospheric chemistry models and their increased ability to simulate regional trends, studies using them choose specific regions. As such, some regions are more frequently represented in the literature than others. Notably, the majority of existing work on future air quality co-benefits of climate mitigation focuses on China and India (Von Schneidemesser et al. 2020). Examples include Kumar et al. (2018) and Chowdhury et al. (2020) for India and Cheng et al. (2021) and Conibear et al. (2022) for China. Other domains studied previously include the Korean peninsula and Japan (Kim et al. 2020) and the USA (Zhang et al. 2017). We chose to focus our domain on Europe as it is an under-represented region in the literature. Although studies that focus on Europe or subregions of exist, they largely use

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CMIP5 emissions instead of CMIP6 (e.g. Fenech et al. 2021 and Sa et al. 2016) or have the primary focus of quantifying the impacts of climate change itself on air quality as opposed to emissions change (Tainio et al. 2013; Tarin-Carrasco et al. 2019).

To summarise, existing research suggests there is additional work needed to understand the impact on future air quality from changes in pollutant emissions associated with the SSPs across Europe using models capable of capturing the changes in air quality at greater spatial resolution. Therefore we present an exploration of the potential mid-century air quality impacts in Europe following the emissions changes from three up-to-date SSPs (SSP1-2.6, SSP2-4.5, and SSP3-7.0) using a state-of-the-art regional atmospheric chemistry model. This aims to help us understand the implications of these updated emissions changes on a sub-regional scale in the European domain.

2 Method

2.1 Model Description

We use the Weather Research and Forecasting coupled with Chemistry model version 4.2. (WRF-Chem). This is an Eulerian, grid-based atmospheric chemistry model. Grell et al. (2005) provide a general model description. We use WRF-Chem at 30 km horizontal resolution with 38 vertical levels up to 50 hPa and a domain of 100*100 grid boxes ranging from latitudes 32° North to 60° North and longitudes from 22° West to 30° East in the North of the domain, narrowing to 13° West to 19° East in the Mediterranean (Supplementary Figure A1). Note that the model domain does not cover all of Europe and for the purpose of this study we define "Europe" as 13 countries: Germany, the UK, France, Spain, Italy, the Netherlands, the Czech Republic, Hungary, Poland, Slovakia, Ireland, Slovenia and Portugal. These countries have a combined population of approximately 380 million and represent a range of sources of primary air pollutants and environmental conditions that will affect air quality.

We simulate a present-day air quality control with gridded 2014 emissions used in CMIP6 (Hoesly et al. 2018) and simulations with anthropogenic emissions representing 2050 for each of SSP1-2.6, SSP2-4.5 and SSP3-7.0 (Feng et al. 2020). The model parameters are shown in Table 1. For all scenarios the meteorology was fixed at 2014 conditions using meteorological initial and boundary conditions from ECMWF ERA5 (Hersbach et al. 2020). 2014 was chosen as this is the most recent year of historical emissions data from the emissions inventory used in CMIP6. We also use CMIP6 output from CESM2-WACCM (Danabosglu, 2019) simulations to provide initial and chemical boundary conditions. To simulate chemistry, a scheme described by Hodzic & Knote (2014) is used that combines MOZART-4 gas phase chemistry, which includes 85 gas-phase species, 157 gas phase reactions and 39 photolysis reactions (this scheme and the included reactions are provided by Emmons et al. 2010) with the MOSAIC aerosol chemistry scheme described initially by Zaveri et al. (2008). This provides detailed chemistry for a range of aerosol species including nitrate from ammonium nitrate (NO_3), sulfate (SO_4), organic carbon (OC), black carbon (BC), ammonium from other sources (NH_4), sodium and chloride, all in four size bins up to 10 μg. The combined scheme described by Hodzic & Knote (2014) enhances these by including aqueous chemistry, improved treatment of monoter-

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penes and hydrocarbons, and updating the mechanism calculating secondary organic aerosols. $PM_{2.5}$ in the model is the sum of the total dry aerosol mass in 3 smallest size bins (up to 2.5 μg) of the above aerosol components and "other inorganics" (OIN) which largely consists of dust. The full range of model inputs are shown in Table 1.

Parameter	Scheme Used	Source
Chemical Scheme	MOZART-MOSAIC 4-bin Aerosol w/Aqueous Chemistry	Hodzic and Knote (2014)
Biogenic Emissions	MEGAN v2.0.4	Guenther et al.(2006)
Fire Emissions	FINNv1.5	Wiedinmyer et al. (2010)
Natural Dust Emissions	GOCART	Chin et al. (2000), Ginoux et al. (2001)
Meteorological Boundary Conditions	ECMWF ERA5	Hersbach et al. (2020)
Chemical Boundary Conditions	CESM2-WACCM CMIP6 Simulations	Danabasoglu (2019)

Table 1. Model parameters used and sources.

2.2 Emissions associated with CMIP6

The emissions scenarios we chose cover a range of futures. SSP1-2.6 represents a scenario with accelerated mitigation of greenhouse gases and sustainable societal development. SSP2-4.5 is a "middle of the road" scenario in which the trajectory of greenhouse gas mitigation and sustainable development does not accelerate or decelerate strongly. SSP3-7.0 is a scenario in which regional rivalry hampers greenhouse gas mitigation and sustainable development. The assumptions in air pollutant controls mirror the trajectories of greenhouse gas emissions in each scenario, with some non-linearity or deviation in particular species to match the scenario narrative. These are explained by Rao et al. (2017); In summary, SSP1 assumes an acceleration in pollution control progress, SSP3 a deceleration, and SSP2 neither a notable acceleration or deceleration from present-day controls. Figure 1 shows how the emissions of key species change in future scenarios compared to the present day, demonstrating how the narrative scenarios translate to emissions data. Here the non-methane VOCs are grouped. We see that SSP1-2.6 has considerably lower emissions of all pollutant species compared to the present day. SSP2-4.5 and SSP3-7.0 have lower reductions in emissions overall, but notably differing trajectories for NH_3 emissions, which increase compared to the present following both scenarios, CH_4 which is mitigated following SSP2-4.5 but worsens following SSP3-7.0 and CO, which is heavily mitigated following SSP1-2.6 and SS2-4.5 but reduces only minimally following SSP3-7.0.

Table 2 shows the European total emissions of air pollutants assumed for the present-day scenario in 2014 and in the scenarios we perform simulations for 2050, taken from the input emissions files from Hoesly et al. (2018) for the present-day and Feng et al. (2020) for the future scenarios. All the emissions files were at 50 km horizontal resolution. The emissions were then

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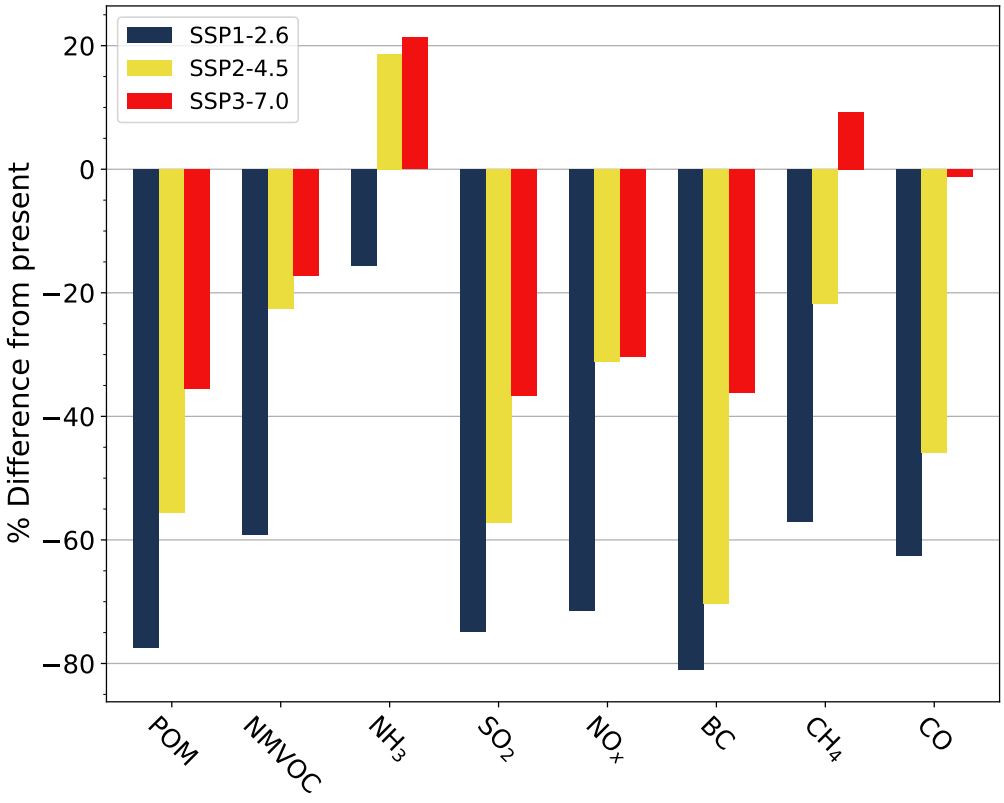

Figure 1. Relative change of model domain average annual emissions from 2014 to each of the future scenarios in 2050.

regridded to 30km.

Pollutant	2015 emissions	SSP1-2.6 (2050)	SSP2-4.5 (2050)	SSP3-7.0 (2050)
Black Carbon	9.7	3.6	6.3	11
CO	934.3	502.8	796.7	996
NH_3	65.3	64.2	76	80.7
NO_x	155.5	74.5	116.8	169.9
SO_2	100.8	26.7	53.3	99.8
Organic Carbon	34.8	18.2	26.9	38.4
NMVOC	227.2	109.1	197.2	256.2
CH_4	388.1	211.1	357.2	559

Table 2. European (domain defined above) total emissions of air pollutants in 2014 from CMIP6 and 2050 from ScenarioMIP SSP1-2.6, SSP2-4.5 and SSP3-7.0, all expressed in Mt/yr

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As the CMIP6 emissions do not include a component of inorganic $PM_{2.5}$ or PM_{10} directly emitted as anthropogenic dust (as required for WRF-Chem) we created files to simulate this fraction using linear regression based on the anthropogenic carbon monoxide emissions using the relationship between EDGAR-HTAPv2 carbon monoxide and $PM_{2.5}$/ PM_{10} (which represent anthropogenic dust) emissions (Janssens-Maenhout et al. 2015). This methodology has been used previously by Kumar et al. (2018) and Wu et al. (2019). These input files are referred to in the rest of the text as anthropogenic dust emissions. To generate emissions of individual non-methane VOC (NMVOC) chemical species we use scaling factors derived from ratio of individual NMVOCs to total NMVOCs in the EDGAR-HTAPv2 emissions inventory (Huang et al. 2017). This provided a greater spectrum of speciated VOCs than the scaling factors used by Hoesly et al. (2018) and Feng et al. (2020).

2.3 Model Output

We use hourly output from each the year-long WRF-Chem simulations for each species (O_3, CO, CH_4, SO_2, NO_2, Nitrogen Oxide (NO), NH_3 and $PM_{2.5}$ dry aerosol mass, and separate files for the individual $PM_{2.5}$ components, NO_3, NH_4, SO_4, OC, BC, Sodium and Chloride). All air pollutant output was analysed only at surface level. For some analysis, we weighted $PM_{2.5}$ and O_3 by population using the formula outlined in Abdul Shakor et al. (2020). We used time-varying gridded population projections for each SSP from Jones & O'Neill (2016). To represent the present-day population, the SSP2 population projection for 2020 was used.

2.4 Model Validation

The present-day simulation for 2014 was validated against $PM_{2.5}$, O_3 and aerosol component observations (as detailed in Table 3) from the European Modelling and Evaluation Programme (EMEP) as this features sites for a range of species across Europe. Coastal sites and sites with an altitude above 1 km were excluded as a model with this resolution would not be expected to simulate air pollutant concentrations at these locations well. We used spatial linear interpolation to extract data from our gridded model output to compare to the locations of the observation sites. The sites used are shown on Supplementary figure A1

Comparisons of modelled and observed O_3 and $PM_{2.5}$ are shown in Figures 2 and 3. Figure 2 shows observed and simulated monthly mean $PM_{2.5}$ and O_3 colour-coded by the observation station. Simulated monthly O_3 data shows a slight underestimation (mean absolute bias compared to observations was -3.40 $\mu g/m^3$) overall. This underestimation is generally larger in the observation sites in Germany (Schmucke, Neuglobsow and Waldhof), however, an overestimation is seen in sites closer to the Mediterranean (Saint-Nazaire, Barcarrota). There were no O_3 observations available for Vredepeel, Cabauw Wielsekade, Guipry and Melpitz. $PM_{2.5}$ showed an overestimation compared to observations, with a mean absolute bias of 7.98 $\mu g/m^3$. The sites with the largest overestimation were Cabauw Wielsekade, Harwell, and Vredepeel. The overestimation was smaller in sites such as San Pablo de los Montes, Barcarrota and Penausende.

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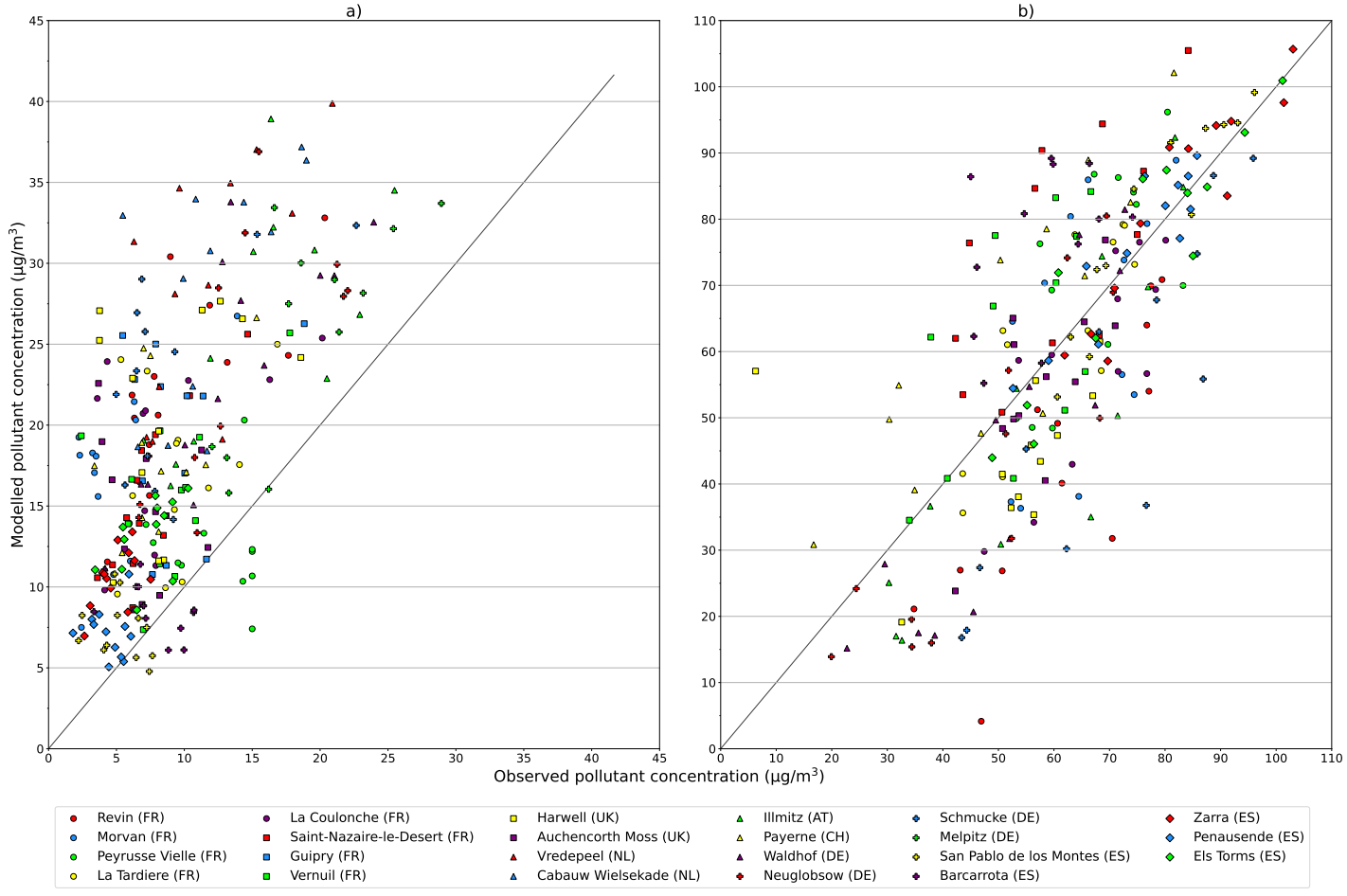

Figure 2. Comparison of modelled $PM_{2.5}$ (a) and O_3 (b) to ground-based Observations. Monthly means are used for both species and the units are $\mu g/m^3$. Model data is interpolated from the latitude and longitude coordinates of the observation site. The middle line on each plot represents what the data would look like if observations and model were equal.

Figure 3 is a comparison of monthly mean modelled data compared to observations (averaged over all sites). The model represented the seasonal cycle, with higher $PM_{2.5}$ in spring and autumn, matching when the emissions peaked. The model $PM_{2.5}$ overestimation was larger in spring and autumn and smaller in summer. Simulated O_3 showed seasonal biases, The model underestimated in winter and spring but overestimated during summer and autumn.

Further comparison was made between the 2014 modelled $PM_{2.5}$ and data from a global gridded $PM_{.2.5}$ reanalysis product created by Van Donkelaar et al. (2021) from a combination of ground-based and satellite observations (Figure 4). The results suggest that as a whole domain average, our $PM_{2.5}$ is higher than the reanalysis, by 4-5 $\mu g/m^3$ in the early months (January-March) before the discrepancy increases over the summer months, due to a larger summertime reduction in $PM_{2.5}$ in the

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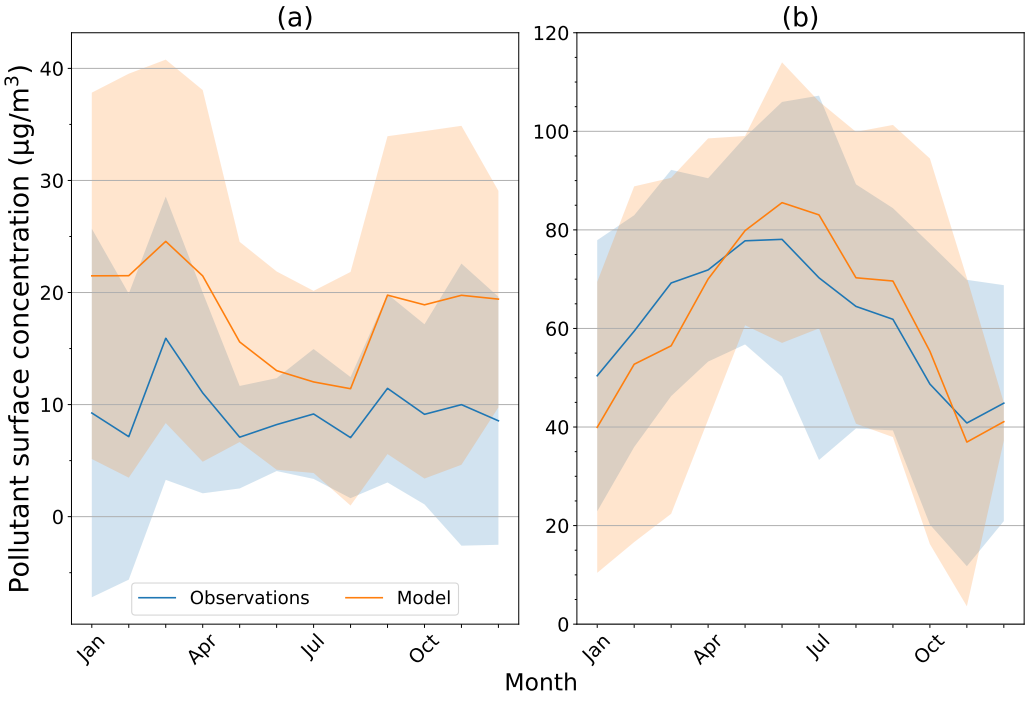

Figure 3. Seasonal cycle in 2014 of a) $PM_{2.5}$ and b) O_3 comparison of monthly mean modelled data to observations (average of all sites). Number of sites = 26 for $PM_{2.5}$ and 21 for O_3. Both pollutants are measured in $\mu g/m^3$. The shaded areas represent variation from the mean across sites as 2* standard deviation.

reanalysis product than in our modelled $PM_{2.5}$. The discrepancy then reduces once again after September (Figure 4a). Much of the $PM_{2.5}$ overestimation over the domain was driven by higher $PM_{2.5}$ in the urban regions of North-Central Europe, including the Rhine-Ruhr, and the Benelux region where the model overestimated observed $PM_{2.5}$ by more than a factor of 2. We do, however, see lower $PM_{2.5}$ than the reanalysis in most of non-coastal Spain, in Poland, and in Balkan countries (Croatia, Bosnia & Herzegovina, Montenegro) (Figure 4b).

Turnock et al. (2020) reported an underestimation of $PM_{2.5}$ compared to observations in Europe in a similar period (2005-2014). This is likely because of the additional emission source of $PM_{2.5}$ in our simulations and the coarse resolution of Turnock et al. (2020)'s simulations. Conversely, $PM_{2.5}$ overestimations have been seen in other studies using CMIP6 emissions to drive regional models, such as Cheng et al. (2021) simulating over China, who found that nitrate, sulfate and ammonium $PM_{2.5}$ were overestimated compared to observations by 30-60%.

Further validation of $PM_{2.5}$ components (comparison of modelled values with ground-based observations) was conducted to diagnose the difference between model and observations. These are shown in Table 3. The total bias in $PM_{2.5}$ is greater than the combined bias of the individual aerosol species. Although not every observation site measured each species and therefore the

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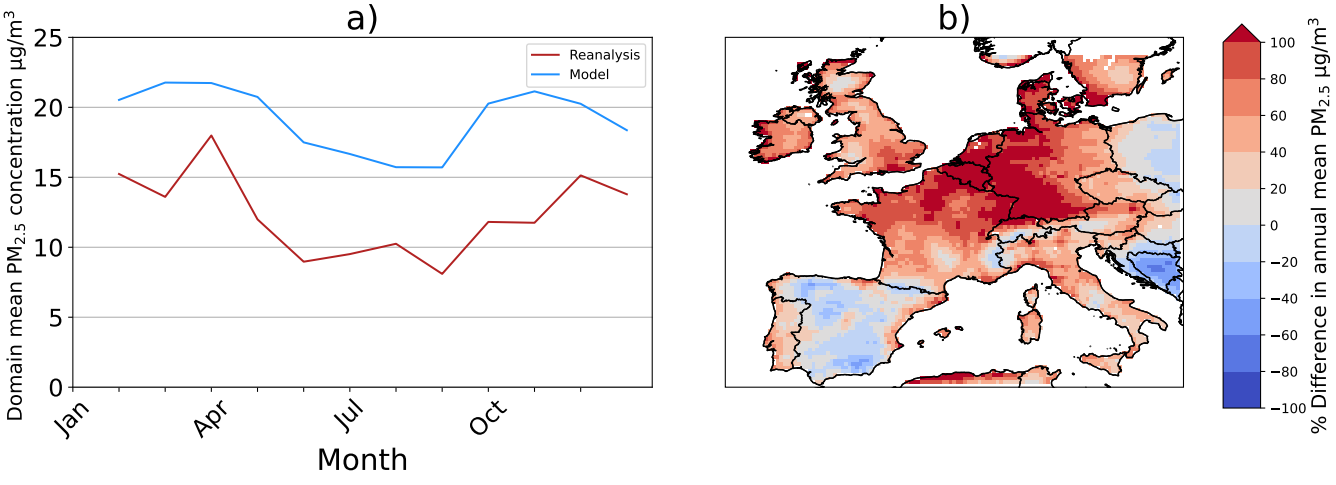

Figure 4. (a) shows a domain average time series of monthly $PM_{2.5}$ data from the present day simulation compared to the domain average of monthly $PM_{2.5}$ from the Van Donkelaar et al. (2021) reanalysis. (b) shows the percentage difference of the annual mean $PM_{2.5}$ from these same data sources, with the reanalysis regridded to the model output for better comparison.

proportions cannot be assumed to be the same at each site, this implies that a different source may account for much of the bias. A large proportion of the bias likely comes from the OIN (dust) component. This agrees with previous research; Im et al. (2015) find that WRF-Chem setups using GOCART over-produce dust and that the partitioning of dust particulates between $PM_{2.5}$ and PM_{10} when MOSAIC aerosols does not reflect real conditions over Europe. Similarly, Georgiou et al. (2018) validating WRF-Chem over Cyprus show that WRF-Chem simulations with MOSAIC aerosols can result in a significant overestimation of $PM_{2.5}$, largely driven by the dust scheme. Additionally, some of the overestimation in the OIN component is likely the result of the derived anthropogenic dust emissions as this is calculated from the CO emissions and it would explain the larger overall $PM_{2.5}$ overestimation in polluted, urban regions.

Table 3 shows that the model overestimates NO_3 aerosol by 2.8 $\mu g/m^3$b and underestimates SO_4 aerosol by -2.1 $\mu g/m^3$ when compared to the observation sites. Overestimation of NO_3 aerosol matches the findings of other WRF-Chem studies including Cheng et al. (2021) and Balzarini et al. (2015), however, both of these studies found SO_4 overestimation as opposed to underestimation. The NO_3 overestimation may be the result of high NH_3 emissions over much of the year for the emissions used in CMIP6 in comparison to other emissions inventories. When compared to EDGAR-HTAPv3 (Crippa et al. 2023) CMIP6 NH_3 emissions were lower during February, March, and April, but higher the rest of the year. Similarly, the CMIP6 emissions of nitrogen oxides (NO_x) are generally higher than EDGAR-HTAPv3 in urban regions, which may also contribute to the overestimation in NO_3 aerosols.

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Pollutant	N (sites)	Model Mean (μg/m^3)	Observations Mean (μg/m^3)	Absolute Mean Difference (model-obs) (μg/m^3)	Root Mean Square error
$PM_{2.5}$	24	18	10	8	9.1
Nitrate (NO_3)	14	5.8	3.1	2.7	4
Ammonium (NH_4)	10	1.5	1.2	0.3	0.6
Organic Carbon (OC)	24	3	2.9	0.1	1.5
sulfate (SO_4)	10	1.3	3.4	-2.1	2.5
Black Carbon	13	0.8	0.4	0.4	0.6
Chloride	13	1.4	0.2	1.2	1.5
Sodium	14	1.9	0.2	1.7	1.9

Table 3. Mean difference between annual mean model output and annual mean ground-based observations for different $PM_{2.5}$ components across monitoring sites in Europe

		Change from present-day in μg/m^3 (% in brackets)		
Domain statistic	**Present day concentration (μg/m^3)**	**SSP1-2.6**	**SSP2-4.5**	**SSP3-7.0**
Minimum	4.9	-1.5 (-31)	-0.61 (-13)	-0.1 (-2)
Mean	19.4	-7.3 (-38)	-4.2 (-21.6)	-2.1 (-10.8)
Maximum	44.4	-21.3 (-48)	-17.6 (-40)	-5.6 (-12.6)
Population-weighted mean	17.5	-8.6 (49)	-5.2 (30)	+0.9 (5)

Table 4. Annual mean $PM_{2.5}$ whole domain change statistics for each future scenario in 2050 compared to the present day baseline (the left-hand column). For future scenarios, the raw change for each of these is shown in μg/m^3 followed by the percentage change in brackets.

3 Results

3.1 Changes in $PM_{2.5}$

Table 4 shows the European annual mean $PM_{2.5}$ in the present day and the change from this in the future scenarios. In general, annual mean $PM_{2.5}$ reduced in all future scenarios compared to the present day. The future reduction in European annual mean $PM_{2.5}$ of 38% in SSP1-2.6 was far greater than the 11% following SSP3-7.0. There are differences in the pattern when population-weighting is applied; overall, population exposure to $PM_{2.5}$ increases slightly following SSP3-7.0 despite the domain-wide decrease. This suggests that the majority of the increases in $PM_{2.5}$ are in highly populated areas

Spatially (Figure 5), we see greater reductions in $PM_{2.5}$ in urban and industrial regions than the domain average. Both industrial and urban regions see strong $PM_{2.5}$ reductions under SSP1-2.6 and SSP2-4.5 (although these are far larger following SSP1-2.6). Conversely, under SSP3-7.0, only urban regions see considerable reductions in $PM_{2.5}$. This would be expected as

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reductions in industrial emissions of NO_x or SO_2 are smaller following SSP3-7.0 and SSP2-4.5. Additionally, under SSP3-7.0, localised areas of worsening air quality are seen, including around East Yorkshire, UK (worsening up to 4 $\mu g/m^3$) and Zeeland and South Holland, the Netherlands (worsening up to 2 $\mu g/m^3$). Some of these localised increases correspond with the locations of major combustion power plants, including Drax (UK) and Belchatow (Poland). This is because the emissions scenarios assume that power generation emissions increase up to mid-century compared to the present day following SSP3-7.0, they drop following SSP2-4.5 but this is approximately half the reduction that is predicted following SSP1-2.6. All scenarios show slightly worsening $PM_{2.5}$ air quality of up to 2 $\mu g/m^3$ near Gijon, Spain.

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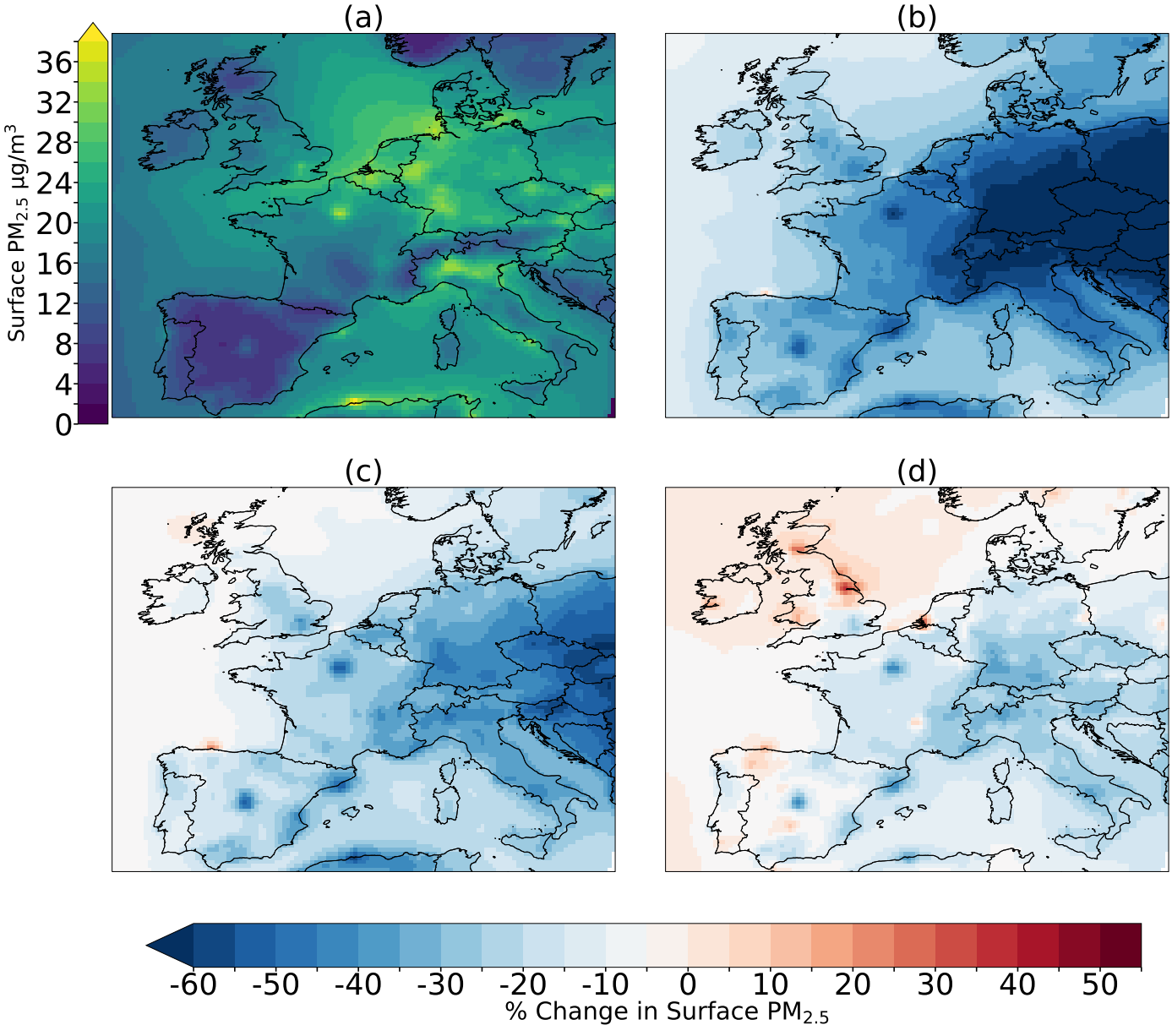

Figure 5. (a) Modelled annual mean $PM_{2.5}$ ($\mu g/m^3$) using 2014 emissions. (b),(c) and (d) are the percentage change from (a) of the annual mean $PM_{2.5}$ ($\mu g/m^3$) simulated using 2050 emissions for SSP1-2.6, SSP2-4.5 and SSP3-7.0 respectively.

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The $PM_{2.5}$ reductions in SSP1-2.6 are larger across most of the domain compared to the other scenarios. This is most notable across Central/Eastern Europe (e.g. Germany, Poland, the Czech Republic, Austria). This is potentially because these regions have a larger proportion of anthropogenic $PM_{2.5}$ sources than natural sources. Smaller improvements are projected in countries such as Portugal and Ireland where natural sources of $PM_{2.5}$ dominate. Additionally, SSP1-2.6 shows large reductions in rural regions compared to the present, whereas these reductions are minimal following SSP2-4.5 and SSP3-7.0. This is likely the result of the reduction in NH_3 emissions in SSP1-2.6, where SSP2-4.5 and SSP3-7.0 have increases in NH_3 emissions (Figure 1). This suggests that the difference in agricultural emissions will be a large driver of the extra reductions in $PM_{2.5}$ following SSP1-2.6 and mitigation of emissions in this sector will be key to achieving improved air quality in Europe.

All countries show overall decreases in population-weighted $PM_{2.5}$ (Figure 6). The magnitude of which varies greatly based on the scenario. Following SSP1-2.6, the percentage decrease ranges from 22.7% in Ireland to 68.6% in Hungary. Other countries with decreases in population-weighted mean $PM_{2.5}$ following SSP1-2.6 greater than 50% include Slovenia, Slovakia and Germany. These countries also see the greatest reductions in $PM_{2.5}$ following the other scenarios, for example the largest reduction following SSP3-7.0 is in Slovenia, at nearly 25%.

In addition to benefiting the least following SSP1-2.6, Ireland benefits the least following SSP2-4.5 with a reduction of 5%. Similar to Ireland, Portugal and Spain do not benefit as much from the emissions changes compared to others. Ireland even shows an increase in population-weighted $PM_{2.5}$ following SSP3-7.0 of nearly 4%, which is also seen in the UK. What this suggests is that the benefits are concentrated in countries where anthropogenic sources dominate $PM_{2.5}$ concentrations in the present day. As coastal island countries, Ireland and the UK likely have a greater proportional quantity of natural sea salt aerosol making up $PM_{2.5}$ and Spain and Portugal are likely to have large proportions of natural dust $PM_{2.5}$ due to proximity to North Africa.

Figure 6 also compares the population-weighted mean to the World Health Organisation annual mean $PM_{2.5}$ guideline value of 5 $\mu g/m^3$. It suggests that following SSP1-2.6, many countries could see $PM_{2.5}$ exposure reduce below interim target values (guidelines the WHO suggest as targets to aim for before reaching the guideline value), representing a significant potential benefit for human health. However, even the emissions reductions from SSP1-2.6 do not result in annual mean population-weighted $PM_{2.5}$ concentrations under this guideline, although the model simulated concentrations might be overestimated. Notably, while countries where $PM_{2.5}$ is dominated by natural sources see less improvement, these have among the lowest $PM_{2.5}$ population exposure in the present day. This means that the benefits of the emissions changes are primarily seen in the countries that most need them. The whole domain average also moves below the WHO interim target 1 of 10 $\mu g/m^3$ following SSP1-2.6, after a reduction of almost 50%.

Figure 7 shows the seasonal cycle of $PM_{2.5}$ components averaged across the entire model domain. SSP1-2.6 has a much lower contribution of anthropogenic $PM_{2.5}$ than SSP2-4.5 and SSP3-7.0, driven by the emissions reductions shown in Figure 2.

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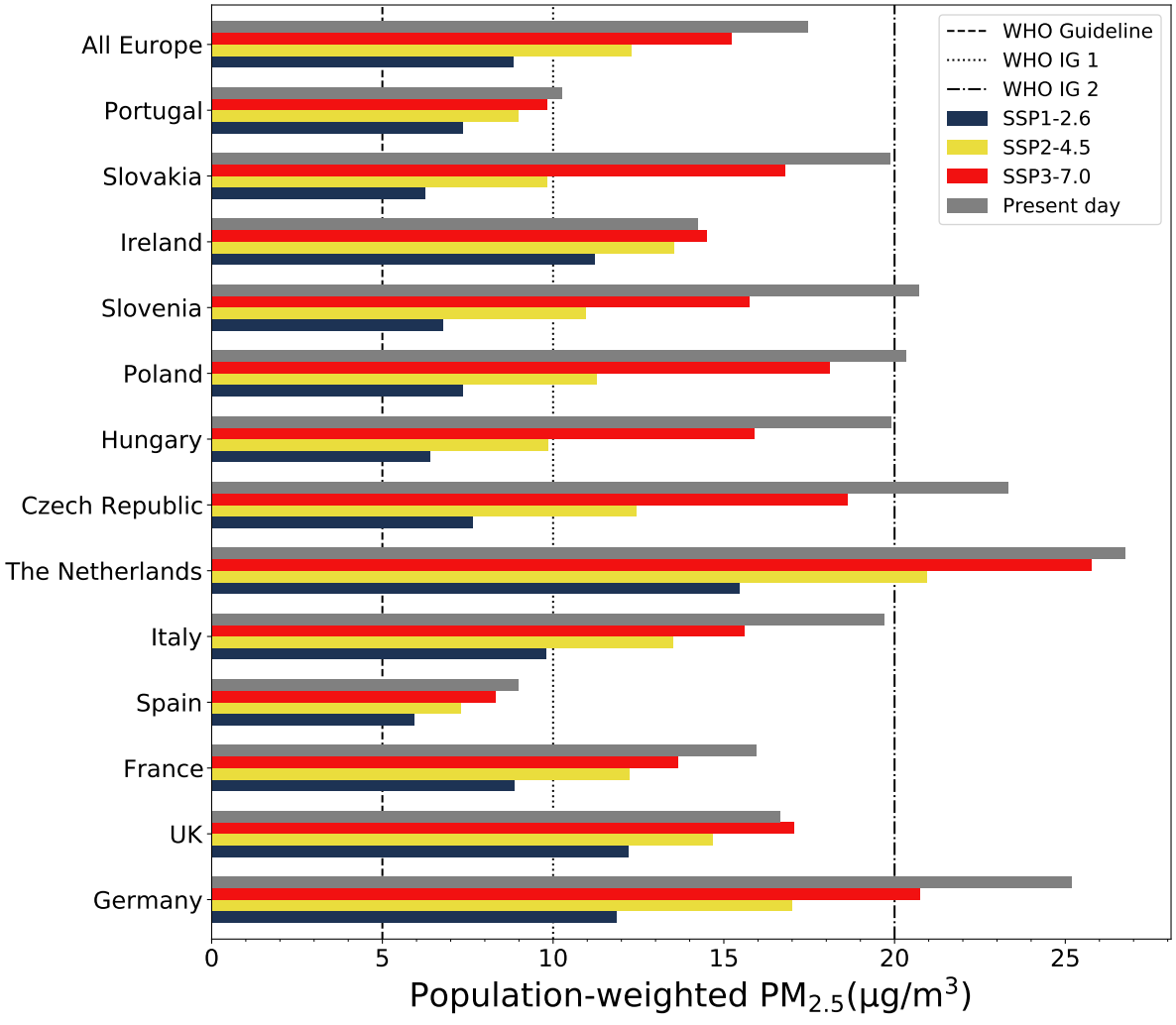

Figure 6. Population-weighted $PM_{2.5}$ in a selection of European countries following different scenarios compared to WHO guideline values. The lines labelled with "IG" are WHO interim guidelines. All Europe refers to a combined population-weighted mean of the 13 countries on the Figure

Figure 7 also shows that it is the change in these anthropogenic species, particularly in NO_3 and OC, that drive the differences between the future scenarios, with NO_3 alone reducing total $PM_{2.5}$ following SSP1-2.6 by over 5μg/m^3 throughout much of the year. The importance of NO_3 aerosol in the future scenarios to determining total $PM_{2.5}$ implies that NH_3 and NO_x emissions reductions will be key to improving future air quality. All future scenarios show overall reductions in NO_x emissions, which can limit the formation of NO_3 and NH_4 particulates (Pusede et al. 2016), however it is only SSP1-2.6 that shows a significant reduction in NO_3 aerosol, likely because it is the only scenario where NH_3 emissions reduce compared to the present day. This suggests that agriculture will be a key sector for attaining air quality co-benefits as agriculture is a major source of NH_3

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320 emissions. The reduction in OC concentrations is proportionally far larger under SSP1-2.6 than the other scenarios, potentially due to the trajectories in power sector OC emissions.

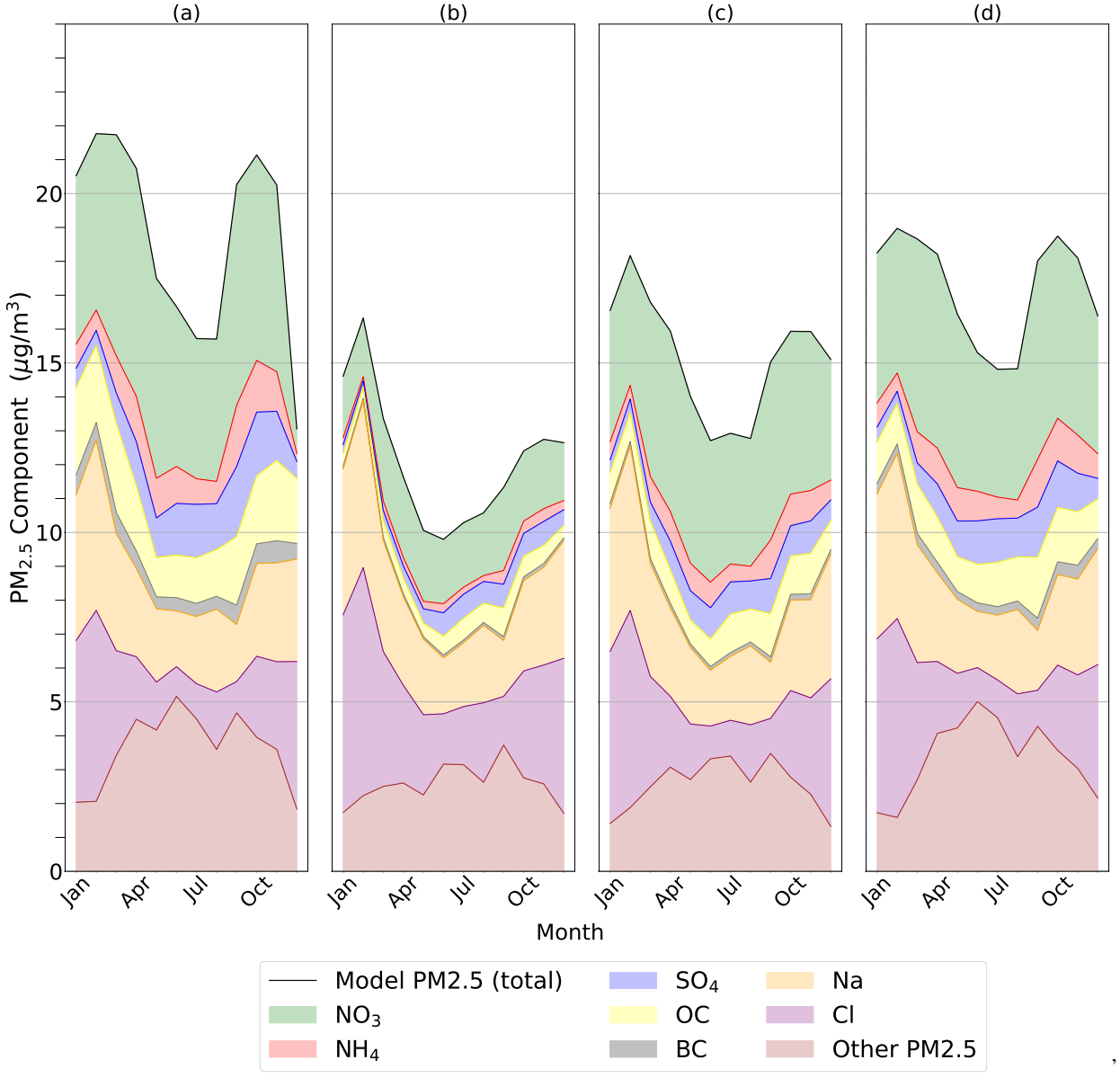

Figure 7. Seasonal cycle of domain average $PM_{2.5}$ over each simulation by component, (a) is the 2014 simulation, (b),(c) and (d) are 2050 from each of SSP1-2.6, SSP2-4.5 and SSP3-7.0 respectively.

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Our findings are in agreement with other work in the area, that air quality co-benefits of climate mitigation are likely for $PM_{2.5}$. When compared to Fenech et al. (2021) for example (who used CMIP5 emissions and focused on the UK), we see that both studies project $PM_{2.5}$ that strong mitigation will result in air quality co-benefits for the UK. The reductions we project for the most comparable scenarios (SSP1-2.6/ RCP 2.6) are larger (-7.3 as opposed to -2.2 $\mu g/m^3$ or approximately 38% vs 25%). We see a diverging trend for the most pessimistic scenarios - while they see reductions in $PM_{2.5}$ concentrations following RCP8.5, we see worsening $PM_{2.5}$ following SSP3-7.0.

Notably, our model simulated high present-day $PM_{2.5}$ in urban regions (e.g.Paris, Madrid and London) compared to surrounding areas. It also produced elevated $PM_{2.5}$ in heavily industrial regions of Europe such as the Po Valley and the Rhine-Ruhr (Figure 5a). It is the changes in these regions that stand out in the other panels of Figure 5. What this suggests, is that our methodology allows us to better represent changes on a local level than work using climate model output. This can be shown when our work is compared to Turnock et al. (2020) (who used CMIP6 output on a global domain) in Figure 8, which shows the difference in change between the present and 2050 following SSP3-7.0 for both $PM_{2.5}$ and O_3. Figure 8 shows that we see similar spatial changes excluding different trends in $PM_{2.5}$ across the Iberian Peninsula and most of the British Isles. This comparison shows how the finer spatial resolution allows us to see localised elevated concentrations of pollution, whereas pollutants are distributed more evenly over the coarser resolution of global models. We also see greater improvements in $PM_{2.5}$ overall than Turnock et al. (2020), for example, for SSP1-2.6 our domain improvement of 7 $\mu g/m^3$ exceeds by more than double theirs of approximately 3 $\mu g/m^3$. This highlights that using air pollutant concentrations from global model simulations may underestimate the extent of future changes in air quality.

Geographically the reductions in European $PM_{2.5}$ are lower than other studies in more polluted regions. Cheng et al. (2021) find a reduction in population-weighted mean $PM_{2.5}$ in China between 2020 and 2050 following SSP1-2.6 of between 20 and 25$\mu g/m^3$ (from approximately 42$\mu g/m^3$ in the present-day scenario). However, this reduction is similar to the average relative reduction of 52% across European countries that we find. Studies on future air quality in India also find that scenarios with a greater focus on sustainability result in reductions in surface air pollution (Chowdhury et al. (2020); Kumar et al.2018) although methodological differences make direct comparisons with these studies challenging. What these comparisons suggest is that Europe could see similar relative air quality co-benefits to other regions following future sustainability scenarios.

3.2 Changes in O_3

Maximum 6-monthly-mean daily-maximum 8h (6mDM8h) O_3 does not show spatial peaks in concentration as clearly as $PM_{2.5}$ does in the present-day simulation (Figure 9), potentially due to inhibited O_3 production in urban areas due to the presence of pollutants such os NO_x and VOCs. The main variation seen is marginally higher concentrations in Mediterranean regions, including Italy, Southern Spain and the French Riviera, which get more sunlight. In the simulations using future scenarios, O_3 largely reduces following SSP1-2.6 (mean reduction of approximately 15%), SSP2-4.5 shows variation across the domain;

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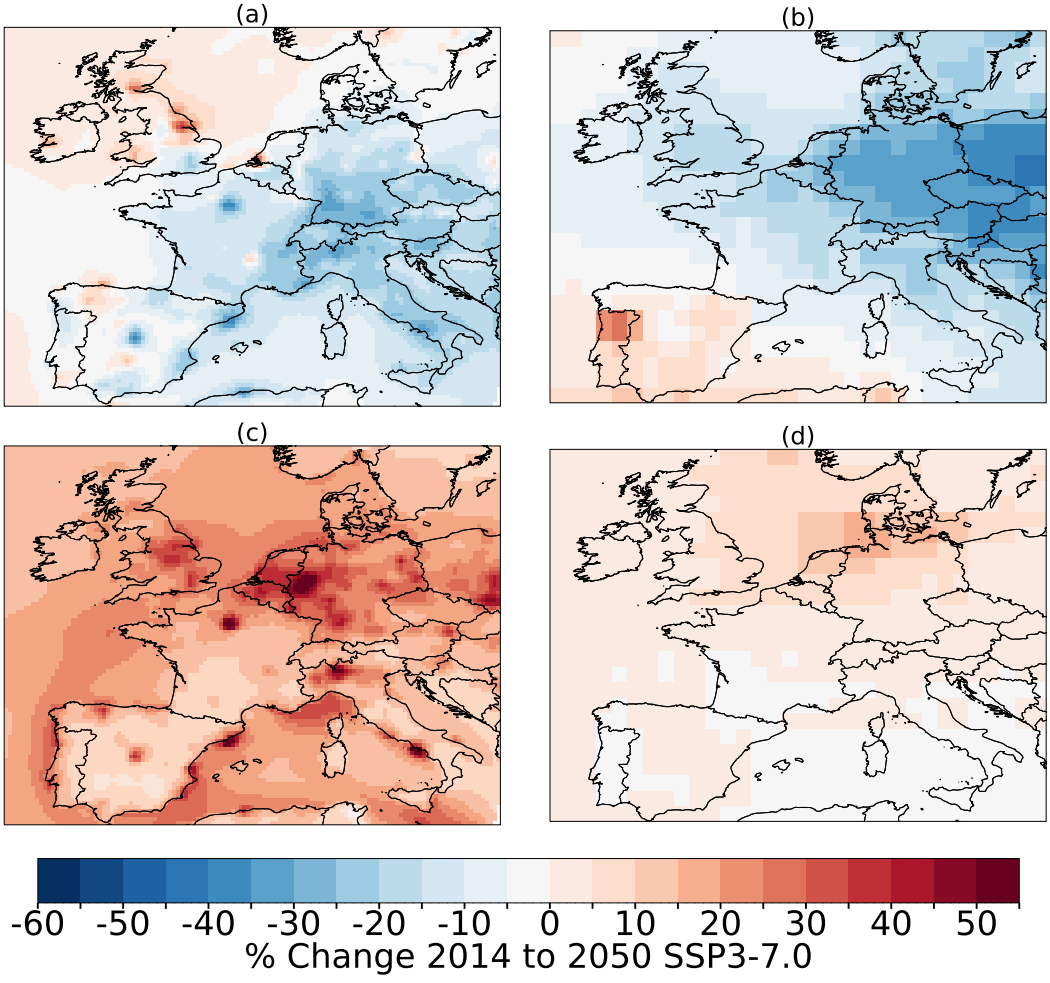

Figure 8. a) Percentage change annual mean $PM_{2.5}$ between present day and 2050 following SSP3-7.0 from our simulations with WRF-Chem b) The same from Turnock et al. (2020) using CMIP6 multi-model output. c) and d) Same as the above, but O_3 compared. Note that this is annual mean O_3 and thus, c) differs from panel d) of Figure 10

while the mean change is a reduction of approximately 3%, increases are seen in most of England, the Benelux region and North-West Germany. 6mDM8h O_3 increases across most of the domain following SSP3-7.0 (mean increase of approximately 13%). This is not universal; small decreases of up to 5% are seen in most of the Mediterranean regions with high present-day O_3. Despite the lack of peaks in the present-day simulation, in the future simulations, O_3 pollution does not reduce as much in urban regions as much of the rest of the domain following SSP1-2.6. The reductions in pollutants that limit formation of O_3 in urban regions may have resulted in these regions not seeing as strong reductions in O_3 (or indeed, increases in O_3) following the future scenarios. The same regions show increases following SSP2-4.5 (and some, such as around Barcelona and Naples show increases following SSP1-2.6) where much of the rest of the domain has reductions in surface level O_3 and the increases

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		Change from present-day in μg/m^3 (% in brackets)		
Domain Statistic	**Present day concentration (μg/m^3)**	**SSP1-2.6**	**SSP2-4.5**	**SSP3-7.0**
Minimum	23.9	+6.4 (+26.7)	+8.3 (+34.7)	+12.3 (+51.5)
Mean	66.2	-9.8 (-15)	+1.7 (+2.6)	+8.7 (+13.1)
Maximum	97.4	-13.3 (-24)	-0.6 (0.6)	+7.6 (+7.8)
Population-weighted mean	52.7	-3 (-7)	+5.3 (+10)	+8.8 (+16.7)

Table 5. Annual mean O_3 whole domain change statistics for each future scenario compared to the present day baseline (the left-hand column). For future scenarios, the raw change for each of these is shown in μg/m^3 followed by the percentage change in brackets.

are higher than surrounding areas following SSP3-7.0. Increases in O_3 following SSP3-7.0 are likely caused by increasing CH_4 emissions, which suggests that agriculture will be a key sector for determining future O_3 pollution. Conversely, as emissions of O_3 precursors including CO and CH_4 decrease compared to the present day following SSP1-2.6 and SSP2-4.5 decrease compared to the present day, the localised increases are expected to be the result of the reduction of an O_3-limiting factor, such as NO_x.//

Annual mean O_3 has a different pattern to 6mDM8h in some scenarios; it is projected to increase universally across the domain following SSP2-4.5, whereas 6mDM8h reduces compared to the present across much of Southern Europe. What this suggests is that there are different regimes across Northern and Southern Europe. The seasonal changes are shown in supplementary Figures A2,A3,A4,A5 and A6. The pattern in 6mDM8h is repeated as expected in June, July and August (Supplementary figure A5), which is likely to be a peak season represented in 6mDM8h. The difference in annual mean compared to 6mDM8h is likely driven by the diverging patterns for Dec, Jan, Feb (Supplementary Figure A3) and Sep, Oct, Nov (Supplementary Figure A6). Sep, Oct Nov shows large increases in O_3 in urban and industrialised regions in all scenarios including SSP1-2.6, suggesting that the reduction in NO_x emissions (which in Europe, usually peak in Autumn and Winter) in all scenarios drive the increases in annual mean O_3 and explain the different pattern from 6mDM8h. In Dec, Jan and Feb, the model produces worsening O_3 concentrations in oceanic regions of Northern Europe, however, this is compared to lower overall O_3 concentrations in the present day compared to the rest of the domain and to other seasons.

The differing trends in CH_4, CO and NO_x (Figure 10) between the scenarios may explain the difference in O_3. It is well established in the literature that in urban areas reductions in NO_x emissions can cause increases in surface level O_3, including within Europe (Lee et al.2020; Finch & Palmer 2020). As the impact of NO_x controls will be greater outside of the summer months this will cause O_3 increases outside of peak season (Supplementary Figure A2). In SSP1-2.6, this effect appears to be masked by much stronger decreases in CO and CH_4 emissions than SSP2-4.5 and SSP3-7.0. What this means is that while all scenarios assume increased pollution control, the additional focus given to climate mitigation (e.g. reducing CH_4 emissions) and the more sustainable socioeconomic development in SSP1-2.6 and SSP2-4.5 has potential air quality co-benefits by outweighing any impacts on O_3 from reduced NO_x. O_3 is considerably more impactful on health during peak season due to the high thresholds needed to affect population health on a large scale, thus most parts of Europe will see reduced impact of O_3

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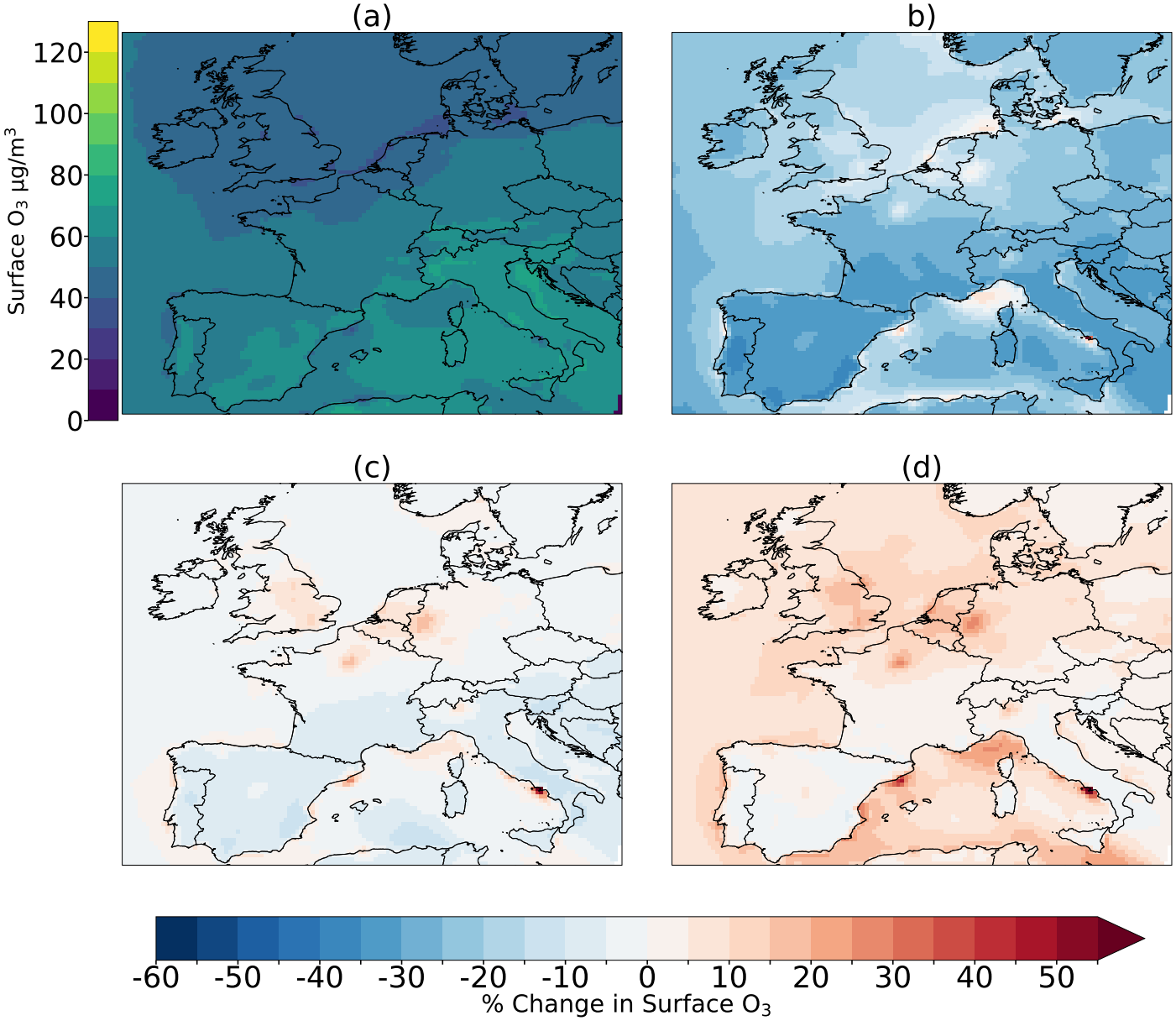

Figure 9. 6mDM8h O_3 calculated as the highest 6-month mean of the highest rolling 8-hour O_3 in 24-hour periods in the O_3 output for each scenario. (a) shows this metric for the CMIP6 2014 simulation. (b),(c) and (d) show the percentage change from this for SSP1-2.6, SSP2-4.5 and SSP3-7.0 respectively.

pollution following both SSP1-2.6 and SSP2-4.5. Reductions in NMVOCs could be having similar effects to the NO_x reductions in some regions. In all scenarios, reductions in NO_x emissions are stronger than NMVOC emissions, however following SSP2-4.5 and SSP3-7.0, the proportional gap is much larger. This suggests that in VOC-limited regimes (which are generally

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urban areas), where reductions in NO_x are more likely to exacerbate O_3, we may see a larger O_3 response. This may account for the larger annual mean O_3 increases in urban regions, especially in SSP2-4.5 and SSP3-7.0. Notably, the emissions changes may cause different patterns in NO_x and VOC-limited regimes that may impact on the O_3 response, for example, Liu et al. (2022) find that in Europe following SSP3-7.0, the percentage of VOC-limited areas drops from nearly 80% to 27% in winter and from 37% to under 3% in summer. If our simulations have a similar change in sensitivity this could suggest that a different precursor, such as CH_4, primarily drives the increases in O_3 following SSP3-7.0.

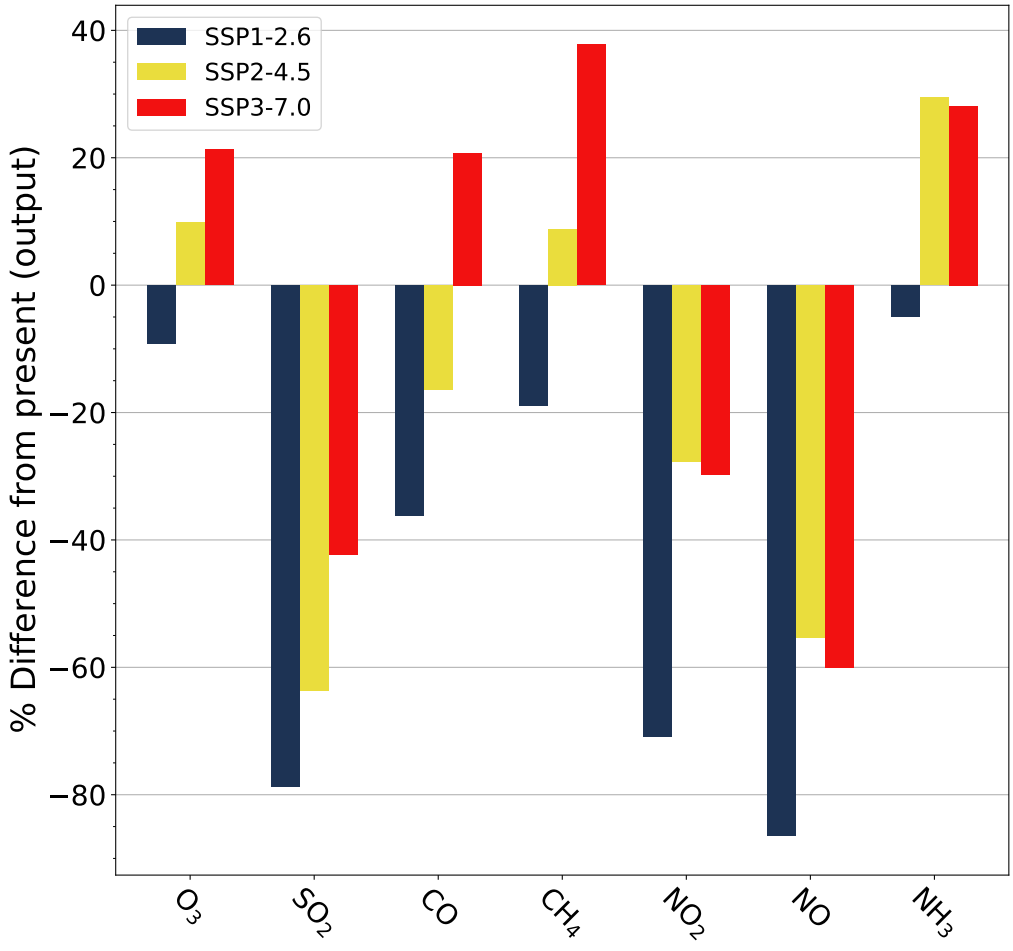

Figure 10. Domain average annual mean O_3 and other air quality relevant species percentage change in the future scenarios compared to the present day.

We find a considerably higher percentage increase in annual mean O_3 in 2050 following SSP3-7.0 compared to that found for Europe by Turnock et al. (2020) (Figure 8). Once again, the difference in resolution is clear here as we see much larger

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increases in O_3 around certain urban regions, whereas Turnock et al. (2020) show a smaller trend distributed over larger areas. As with $PM_{2.5}$, the finer resolution we use could prove valuable, especially for understanding health impacts and trends in urban areas. As shown by the contrasting trends in urban and rural areas for O_3 pollution following some scenarios, being able to represent these changes is valuable.

Compared to studies focusing on regions outside of Europe, our findings are similar to those reported by Zhang et al. (2017) who use WRF-Chem to simulate the impact of RCPs on air pollutants over the USA. They find overall small decreases in 6mDM8h O_3 (up to 2 ppb) except in some large urban areas following RCP 4.5, where they report increases of up to 10 ppb. Following SSP2-4.5, we see a similar trend (average reduction of -1.5 ppb, but localised increases around some cities and industrialised areas).

4 Summary and Conclusions

We use emissions data for three SSPs (SSP1-2.6, SSP2-4.5 and SSP3-7.0), representing very different climate futures, to simulate air quality in Europe in 2050 compared to the present day. This work uses WRF-Chem v4.2 with a much more detailed chemistry scheme and finer grid resolution than much of the previous work using SSPs to provide a more detailed assessment of potential air quality co-benefits on a regional scale.

We show that $PM_{2.5}$, while expected to reduce compared to the present day across most of Europe in all future scenarios, shows by far the biggest reductions in scenarios with a greater focus on sustainability, and therefore more stringent emissions reductions. We find that in 2050, following SSP1-2.6, mean population-weighted $PM_{2.5}$ concentrations across European countries reduces by 52% compared to 2014. Whilst under SSP2-4.5, this average reduction is 34%. The smallest average reduction was 18% by following SSP3-7.0. The additional benefits we see from following SSP1-2.6 are likely due to emissions reductions in the agricultural and industrial sectors.

We also show a different sign of change O_3 across the scenarios, demonstrating that in the more sustainable scenario, SSP1-2.6 (and to a lesser extent, SSP2-4.5), much of Europe will see reduced 6mDM8h O_3 concentrations, whereas 6mDM8h O_3 will worsen following SSP3-7.0. This is likely driven by a combination of reduced NO_x and increased CH_4 emissions. This demonstrates the importance of reducing CH_4 alongside other O_3 precursor species to avoid reducing the efficacy of overall air pollutant controls, caused by focusing entirely on $PM_{2.5}$ and NO_x without also considering the impacts on O_3, as evident from the increases in O_3 concentrations during the COVID-19 lockdowns where large reductions in NO_x emissions occurred with smaller or no effect on CH_4 (Jephcote et al. 2021; Miyazaki et al. 2021).

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We find that using a regional atmospheric chemistry model provides us with the ability to analyse in more detail where air quality in Europe could change in response to the scenarios and that the patterns in air quality changes using this methodology differ from what you get using climate model output. From that, we can make a more informed hypothesis as to why air pollutants respond the way they do based on sector-specific emissions changes. We demonstrate the value that can be added using this methodology, for example by providing country-specific population-weighted mean changes, which may be more useful to regional and national policymakers. This demonstrates the importance of a combined approach to modelling air quality co-benefits using both global and regional models.

To conclude, our results suggest that air quality co-benefits will be seen if society follows a pathway in which environmental sustainability is a priority, particularly in terms of mitigating climate change. This implies there are potential public health benefits, although the results of this may differ from those of other studies, so further studies to calculate the health benefits are important.

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5 Data Availability

Data is available on an online repository at Zenodo (DOI: 10.5281/zenodo.10781398). Data can alternatively be accessed by contacting Connor Clayton

Appendix A: Supplementary Figures

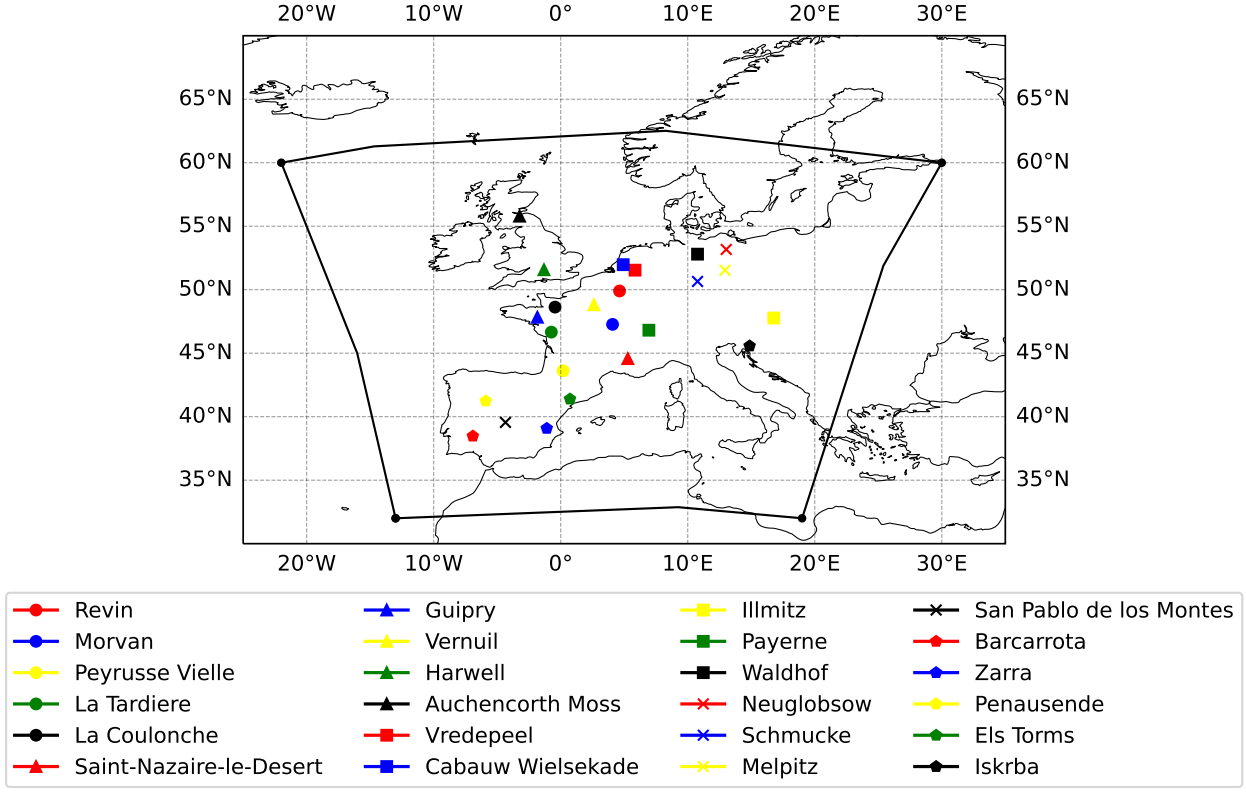

Figure A1. The domain input into WRF-Chem v4.2 for our simulations at 30 km resolution. The observation sites used for model validation are also shown.

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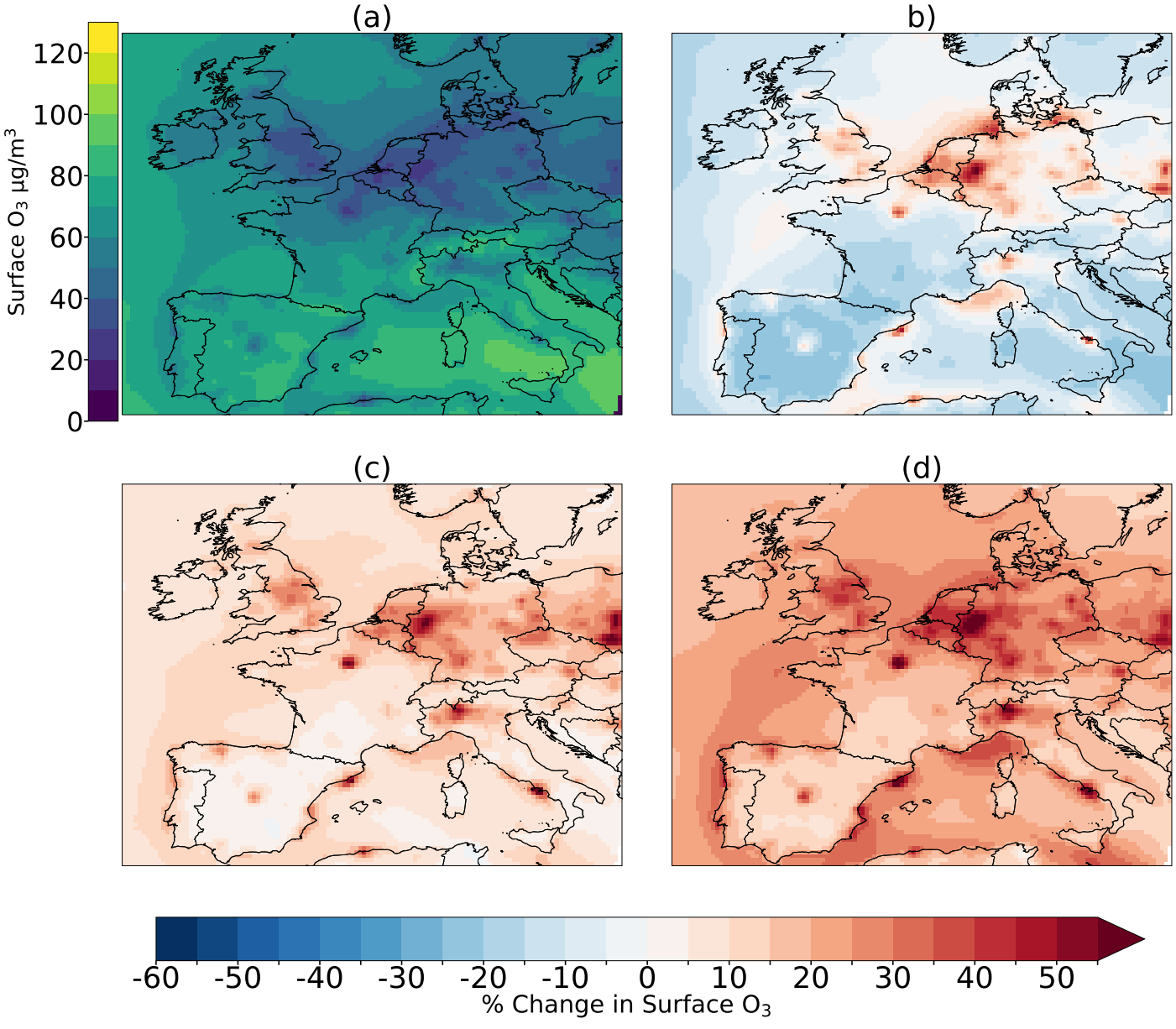

Figure A2. Annual mean O_3 for each scenario. (a) shows this metric for the CMIP6 2014 simulation. (b),(c) and (d) show the percentage change from this for SSP1-2.6, SSP2-4.5 and SSP3-7.0 respectively.

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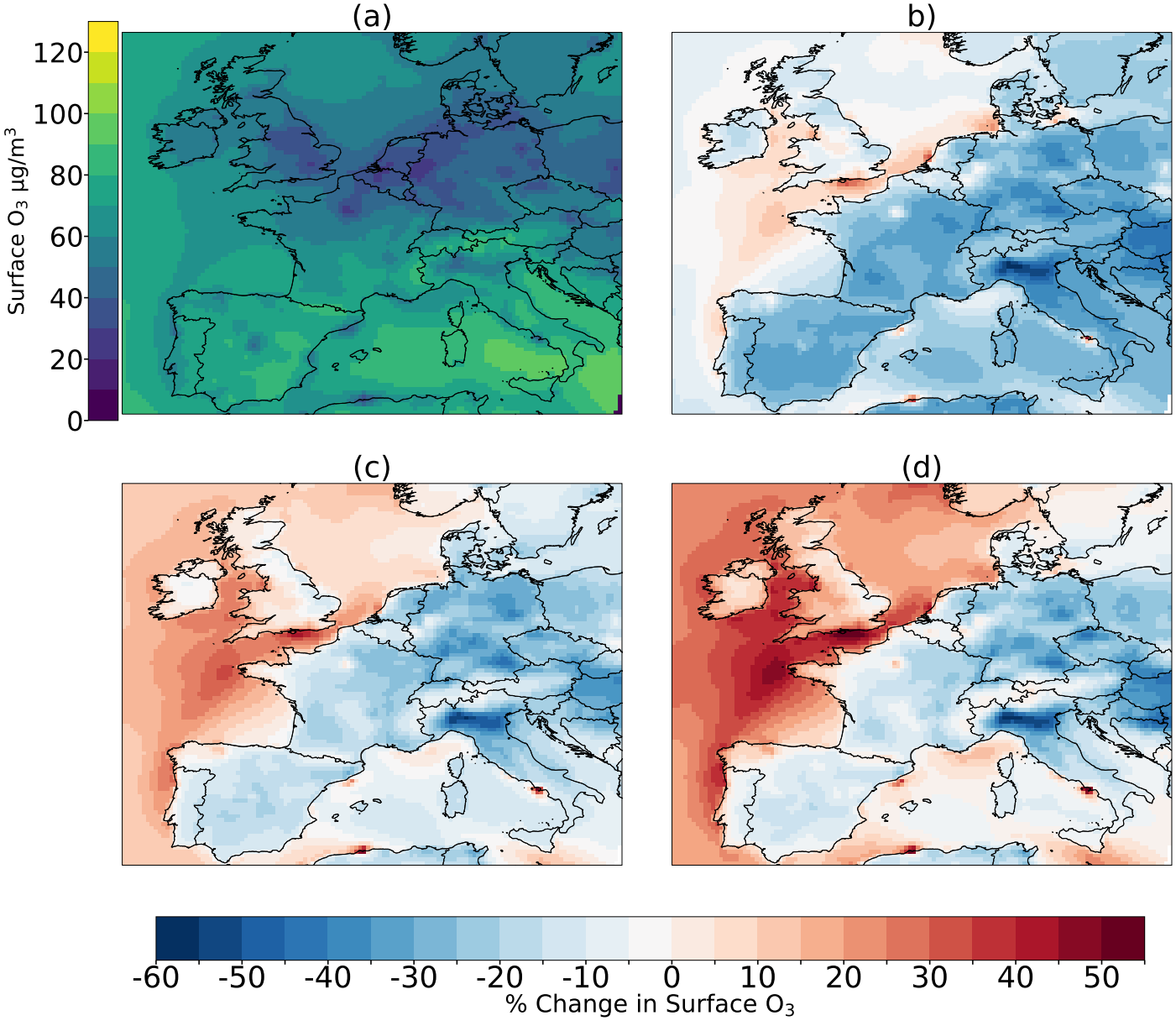

Figure A3. Mean O_3 in January, February and December for each scenario. (a) shows this metric for the CMIP6 2014 simulation. (b),(c) and (d) show the percentage change from this for SSP1-2.6, SSP2-4.5 and SSP3-7.0 respectively.

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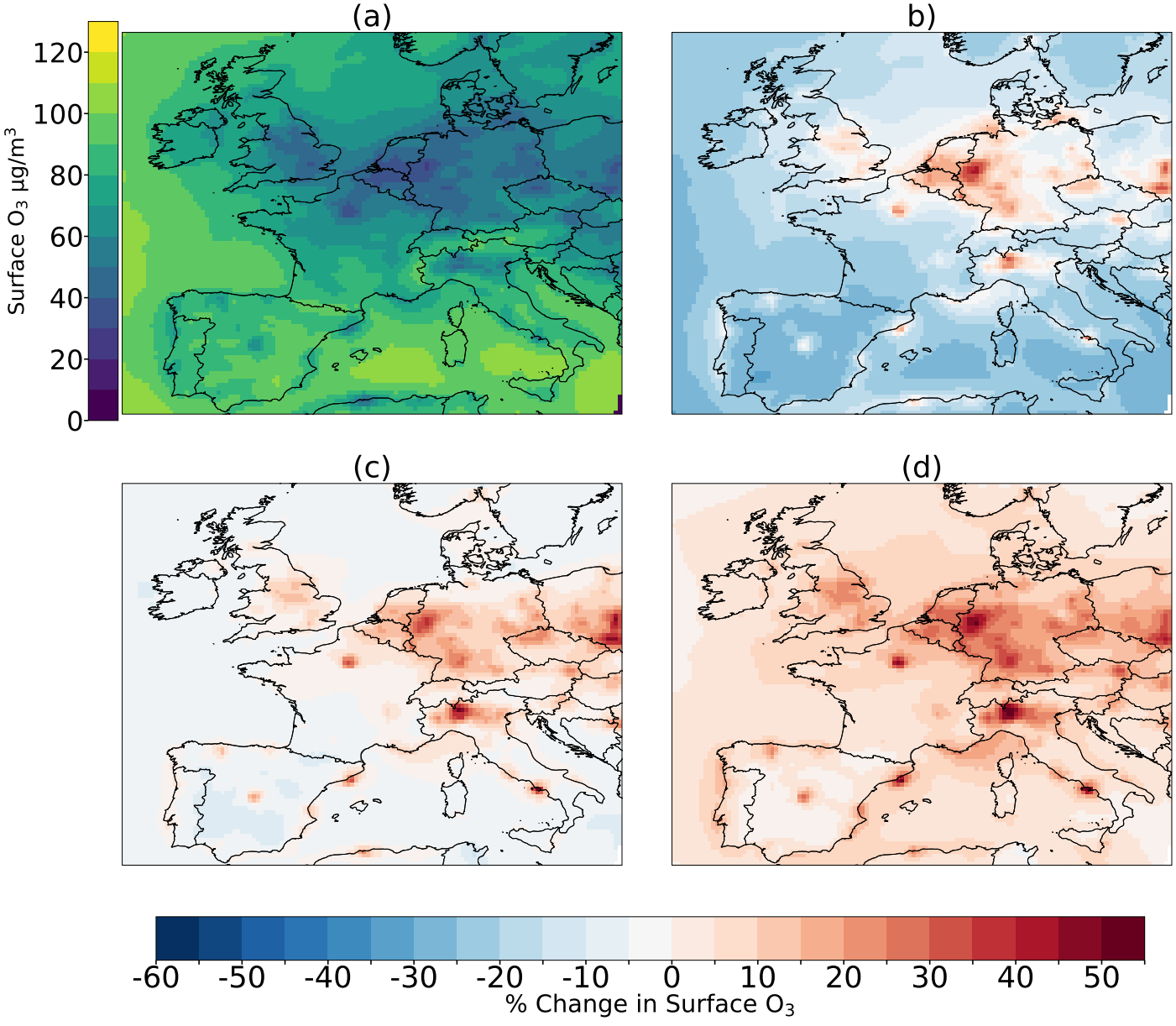

Figure A4. Mean O_3 in March, April and May for each scenario. (a) shows this metric for the CMIP6 2014 simulation. (b),(c) and (d) show the percentage change from this for SSP1-2.6, SSP2-4.5 and SSP3-7.0 respectively.

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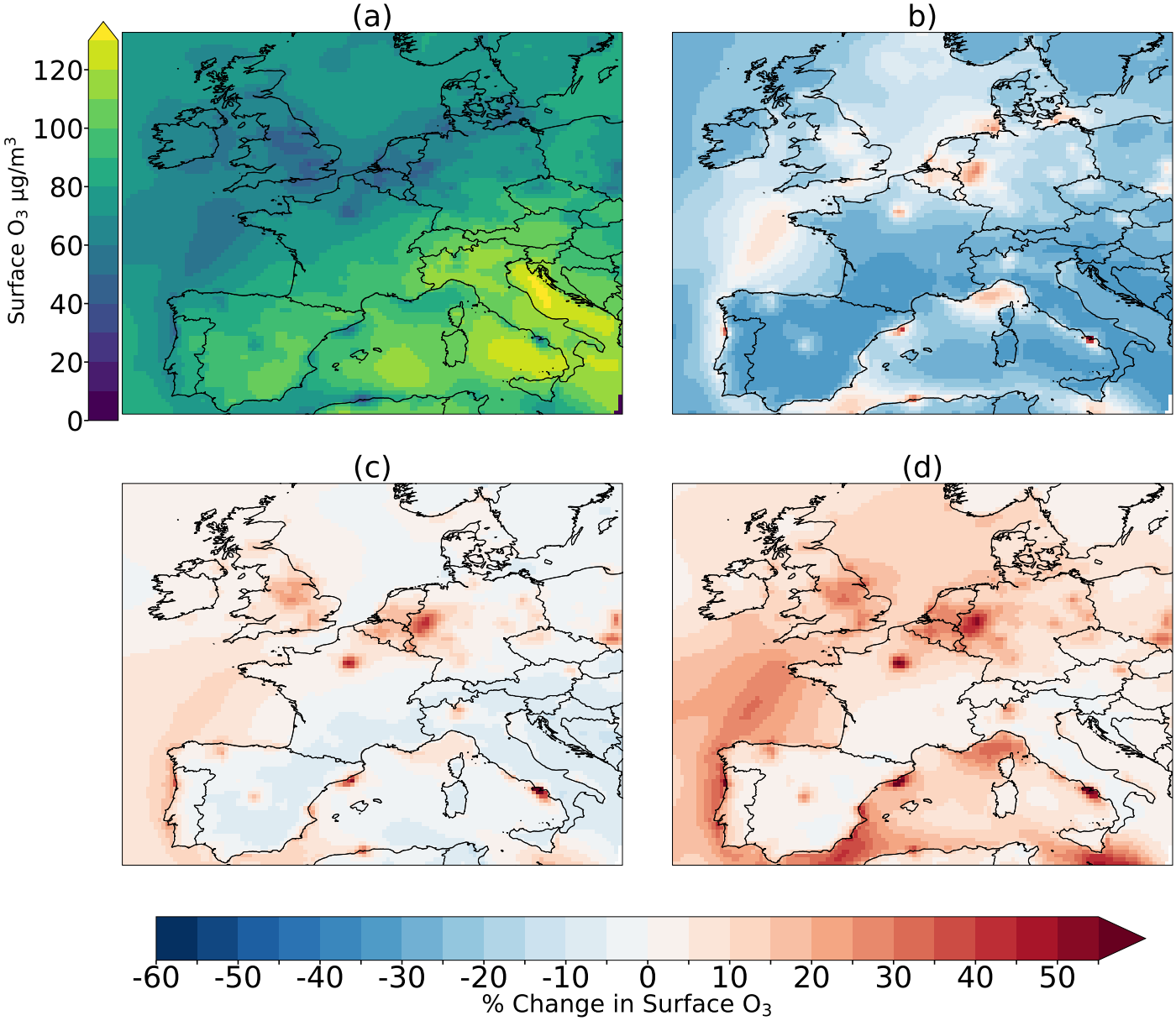

Figure A5. Mean O_3 in June, July and August for each scenario. (a) shows this metric for the CMIP6 2014 simulation. (b),(c) and (d) show the percentage change from this for SSP1-2.6, SSP2-4.5 and SSP3-7.0 respectively.

https://doi.org/10.5194/egusphere-2024-755
Preprint. Discussion started: 21 March 2024

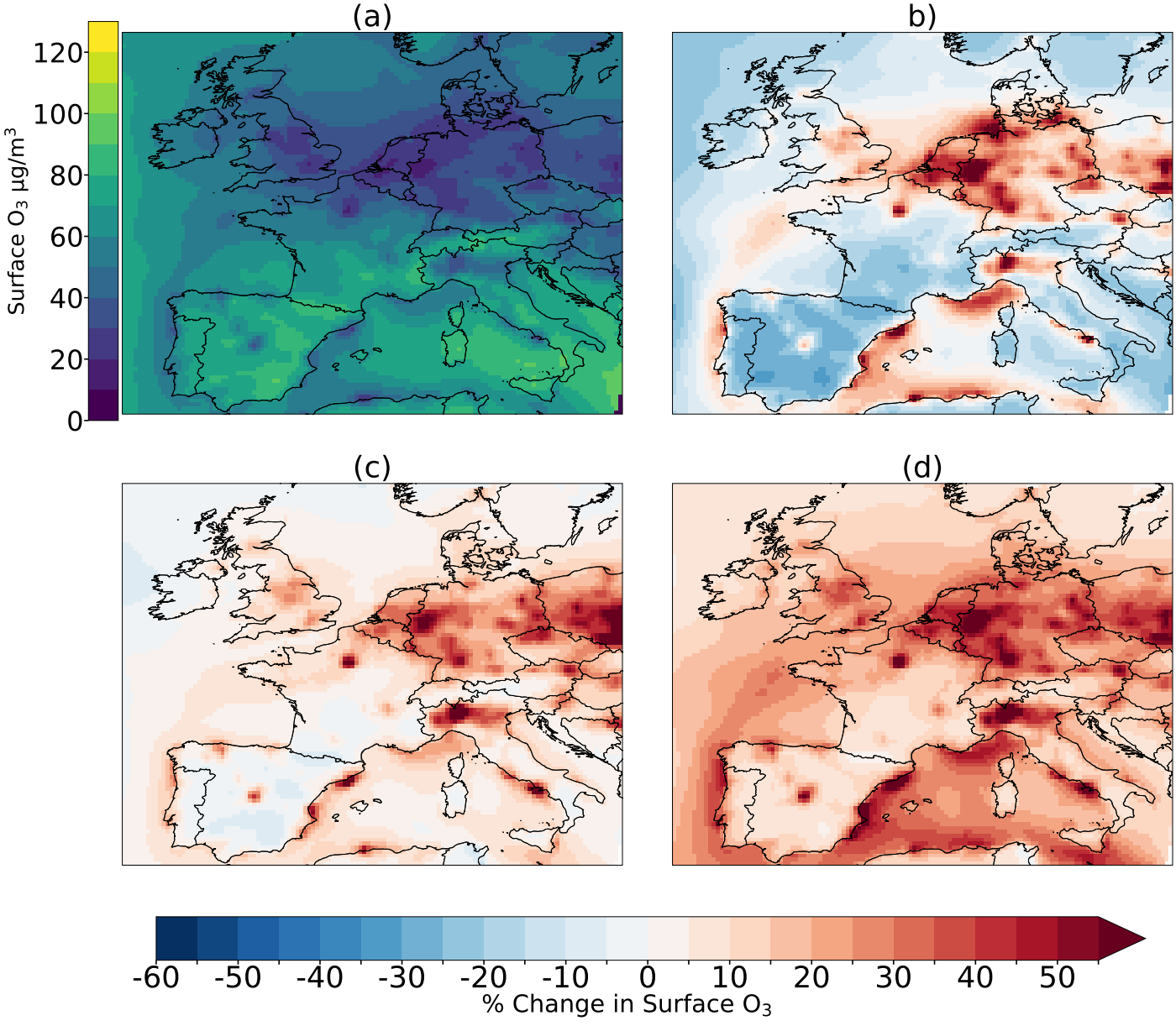

Figure A6. Mean O_3 in September, October and November for each scenario. (a) shows this metric for the CMIP6 2014 simulation. (b),(c) and (d) show the percentage change from this for SSP1-2.6, SSP2-4.5 and SSP3-7.0 respectively.

https://doi.org/10.5194/egusphere-2024-755
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Author contributions. Connor J. Clayton: Performing model simulations, transformed emissions files to work with WRF-Chem, creating the figures and writing the paper.

James B. McQuaid, Daniel R. Marsh, Steven T. Turnock and Kirsty J. Pringle: Devising the main conceptual ideas, supervising project and advising on developing methodology, writing, and interpreting results. CESM2-WACCM boundary conditions were developed and provided by Daniel R. Marsh. Steven T. Turnock provided model output from previous work to compare with these results.

Ailish M. Graham: Assisting with model setup and code to produce some figures. developed code to make chemical boundary conditions work for WRF-Chem.

Carly J. Reddington: Advising on optimising WRF-Chem setup, population-weighting, use of the reanalysis product and interpreting results.

Rajesh Kumar: Advising on transforming SSP emissions to work with WRF-Chem and method of using linear regression to estimate primary $PM_{2.5}$ emissions.

Competing interests. The authors declare that they have no conflict of interest

Acknowledgements. We would like to acknowledge Lansinoh Laboratories Inc. and The Priestley Centre for Climate Futures at the University of Leeds for funding this work and the Met Office for additional funding, supervision and advice. We would also like to thank the Centre for Environmental Modelling and Computation at the University of Leeds for maintaining WRF-Chem at Leeds and providing assistance with implementing the changes to the model inputs needed for this project. We would also like to acknowledge the use of University of Leeds Advanced Research Computing (ARC4) facilities for all model simulations that were used in this project. Rajesh Kumar's contribution to this study is based upon work supported by the NSF National Center for Atmospheric Research, which is a major facility sponsored by the U.S. National Science Foundation under Cooperative Agreement No. 1852977. This work was supported by the UK Natural Environment Research Council (NERC), Grant No. NE/T010401/1.

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