Future impacts of O_3 on respiratory hospital admission in the UK ~~health~~ from current emissions policies

Helen L. Macintyre[a,b], Christina Mitsakou[a], Massimo Vieno[c], Mathew R. Heal[d], Clare Heaviside[e], Karen S. Exley[a,f]

a. UK Health Security Agency, Chilton, Oxon, OX11 0RQ, UK.

b. School of Geography, Earth and Environmental Sciences, University of Birmingham, Edgbaston, B15 2TT, UK.

c. UK Centre for Ecology & Hydrology, Bush Estate, Penicuik, Midlothian, EH26 0QB, UK.

d. School of Chemistry, University of Edinburgh, Joseph Black Building, David Brewster Road, Edinburgh, EH9 3FJ, UK.

e. Institute for Environmental Design and Engineering, University College London, Central House, 14 Upper Woburn Place, London, WC1H 0NN, UK.

f. Department of Health Sciences, University of Leicester, Leicester, UK

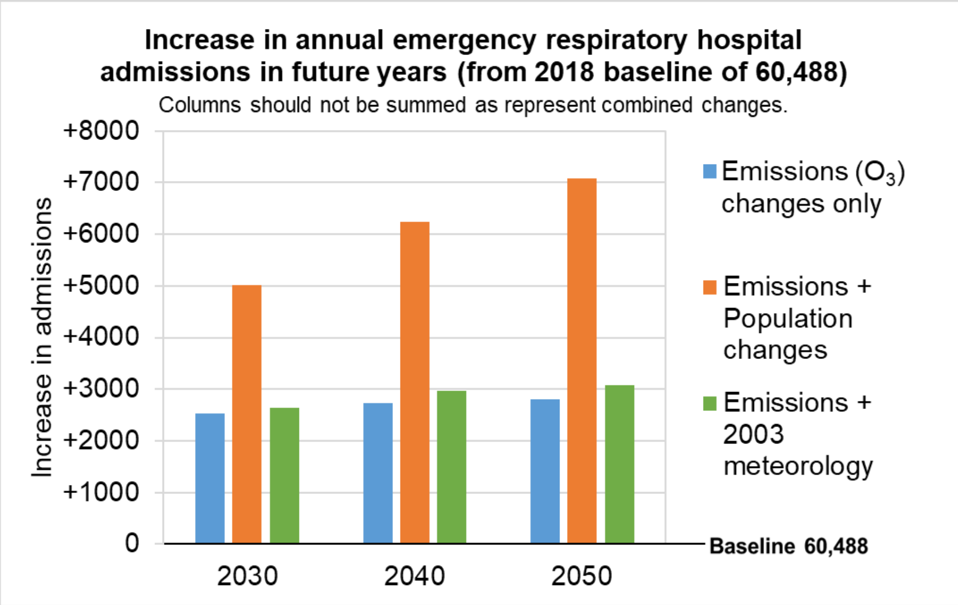
Corresponding author: Dr Helen L Macintyre helen.macintyre@ukhsa.gov.uk

Co-author contacts: Christina Mitsakou christina.mitsakou@ukhsa.gov.uk; Massimo Vieno mvi@ceh.ac.uk; Mathew Heal m.heal@ed.ac.uk; Clare Heaviside c.heaviside@ucl.ac.uk; Karen Exley karen.exley@ukhsa.gov.uk.

Highlights

- Policies on anthropogenic emissions dominate health impacts of air pollution
- O_3 simulated at 3 km resolution for 2030, 2040 and 2050 emissions pathways
- Hospitalisations calculated with latest UK health impact assessment recommendations
- 7078~~Small~~ increase in respiratory hospitalisations by 2050 for current policy c.f. 2018
- Estimated increases are 2-3 times lower if population growth is not included

Graphical abstract



Keywords

Air quality; health impact assessment; emissions; respiratory hospital admissions; ozone

Abstract

Exposure to ambient ozone (O_3) is associated with impacts on human health. O_3 is a secondary pollutant whose concentrations are determined inter alia by emissions of precursors such as oxides of nitrogen (NO_x) and volatile organic compounds (VOCs), and thus future health burdens depend on policies relating to climate and air quality. While emission controls are expected to reduce levels of $PM_{2.5}$ and NO_2 and their associated mortality burdens, for secondary pollutants like O_3 the picture is less clear. Detailed assessments are necessary to provide quantitative estimates of future impacts to support decision-makers. We simulate future O_3 across the UK using a high spatial resolution atmospheric chemistry model with current UK and European policy projections for 2030, 2040 and 2050, and use UK regional population-weighting and latest recommendations on health impact assessment to quantify respiratory emergency hospital admissions associated with short-term effects of O_3. We estimate 60,488 admissions in 2018, increasing by 4.2%, 4.5% and 4.6% by 2030, 2040 and 2050 respectively (assuming a fixed population). Including future population growth, estimated emergency respiratory hospital admissions are 8.3%, 10.3% and 11.7% higher by 2030, 2040 and 2050 respectively. Increasing O_3 concentrations in future are driven by reduced nitric oxide (NO) in urban areas due to reduced emissions, with increases in O_3 mainly occurring in areas with lowest O_3 concentrations currently. Meteorology influences episodes of O_3 on a day-to-day basis,

although a sensitivity study indicates that annual totals of hospital admissions are only slightly impacted by meteorological year. While reducing emissions results in overall benefits to population health (through reduced mortality due to long-term exposure to $PM_{2.5}$ and NO_2), due to the complex chemistry, as NO emissions reduce there are associated local increases in O_3 close to population centres that may increase harms to health~~should be considered in future health advice and planning~~.

1 Introduction & background

Air pollution has detrimental impacts on human health. The main air pollutants associated with impacts on human health are $PM_{2.5}$ (particulate matter with aerodynamic diameter <2.5 μm), nitrogen dioxide (NO_2), and ozone (O_3), with long-term exposure to ambient $PM_{2.5}$ and NO_2 in the UK estimated to have an effect equivalent to 29,000–43,000 deaths annually (Mitsakou et al., 2022). Exposure to ambient O_3 is also associated with ill health, particularly respiratory effects including exacerbation of asthma symptoms.~~;~~ O_3 is estimated to contribute to over 5 million disability adjusted life years (DALYs) lost annually globally (Forouzanfar et al., 2015). The evidence of O_3 impacts on health is strongest for associations between short-term exposure (as quantified using the daily maximum 8-hour running mean O_3) and respiratory and cardiovascular hospital admissions, though with more uncertainty for the latter (COMEAP, 2015). Quantification of the impact on health of long-term exposure to O_3 is not recommended by the UK's Committee on the Medical Effects of air Pollutants (COMEAP) as there is currently insufficient evidence (COMEAP, 2015; Huangfu and Atkinson, 2020; COMEAP, 2022).

Ozone is not directly emitted but is a secondary pollutant formed through complex (photo)chemical reactions in the atmosphere involving NO_x (= NO + NO_2) and volatile organic compound (VOC) emissions, including of methane (CH_4). The levels of methane contribute to the longer-term (annual average) concentrations of O_3 that are also part of O_3 exposure and upon which the shorter-term variations in O_3 driven by non-methane VOCs are superimposed. Concentrations of O_3 are also affected by meteorology, which influences its build-up through photochemistry and air mass transport and dilution, and also via the rate of O_3 deposition to the surface (Royal Society, 2008). High levels of NO_x (e.g. close to emission sources) also act to limit O_3 formation, which is why reductions in NO_x emissions in urban areas, largely associated with improvements in vehicle exhaust emission standards, have led to modelled local increases in O_3 (Carnell et al., 2019).

Air pollution levels have generally improved in the UK over the past several decades, thanks to environmental and public health initiatives. Carnell et al. (2019 reported that reductions in air pollution levels between 1970 and 2010 led to a reduction in UK attributable mortality associated with exposure to $PM_{2.5}$ and NO_2 of 56% and 44% respectively; however, the same study reported an increase in O_3-attributable respiratory mortality of 17%. This reflects a different trend in response to recent air pollutant emissions changes for O_3 in urban areas, where most people live, than for urban NO_2 and $PM_{2.5}$ (AQEG, 2021). In particular, as noted above, there is potential for local increases in O_3 due to reductions in emissions of NO, with a recent report noting upwards trend in urban ozone of 5 – 9 µg m^{-3} between 2000-2019 (AQEG, 2021). A modelling study on possible UK net-zero policies showed that while longer-term (April to September) mean O_3 concentrations may reduce, short-term O_3 metrics revealed less O_3 suppression from reduced NO_x, particularly in winter (Williams et al., 2018). The effect of reduced traffic NO_x emissions increasing urban O_3 was also illustrated during the travel restrictions implemented as part of COVID-19 'lockdown' restrictions (AQEG, 2020; Lee et al., 2020; Jephcote et al., 2021).

Changes in climate will also modify concentrations of surface O_3, as it will for other air pollutants (von Schneidemesser et al., 2015; Doherty et al., 2017). However, the Doherty et al. review concludes that for O_3 over Europe in the next few decades, changes in anthropogenic emissions of the VOC and NO_x precursors will have much greater influence on O_3 levels than changes in climate. For example, model simulations of impacts of climate scenarios and compatible air pollutant emissions under CMIP5 scenarios showed O_3 exposure metrics over Europe in 2050 change by 3% (±8%) to 5% (±11%) due to changes in climate, but by −24% (±10%) to −43% (±7%) due to emission changes (Colette et al., 2013). Over the UK, changes in annual mean O_3 by 2030 (compared to 2003) due to changes in precursor emissions (−3.0 to +3.5 ppbv) are larger than for a +5°C scenario (+1.0 to +1.5 ppbv) including temperature effects on biogenic VOCs, chemical reaction rates and O_3 dry deposition, with the latter being comparable to the changes in O_3 arising from normal interannual meteorological variability (Heal et al., 2013). Resulting changes in mortality burdens were 4.1% associated with O_3 changes due to +5°C, whereas they were 15.6% to 27.7% for emissions changes (depending on emission scenario), assuming no threshold for effects (Heal et al., 2013).

In a previous study we performed an up-to-date quantitative impact assessment of the mortality burdens associated with long-term effects of $PM_{2.5}$ and NO_2 in 2030, 2040 and 2050 using a high spatial resolution atmospheric chemistry and aerosol model with the latest official UK and European emissions datasets (Macintyre et al., 2023). We found that mortality burdens associated with these two pollutants were reduced by over 30% beyond 2030. As O_3 is also impacted by such emission policies, but in more complex ways as described above, we expand our analysis here to consider impacts on respiratory hospital admissions associated with short-term exposure to O_3, using a similar health impact assessment (HIA), to more fully capture future impacts to health associated with changing emissions.

2 Methods

2.1 Air pollution modelling and emissions

Simulations of UK present-day and future air quality at 3 km × 3 km spatial resolution and hourly temporal resolution were undertaken using the EMEP4UK[1] atmospheric chemistry transport model version rv4.36. The chemical model was driven by hourly meteorology from the Weather Research and Forecasting (WRF) model version 4.2.2 (wrf-model.org). The EMEP4UK-WRF modelling system has been widely used to simulate historic, present-day and potential future air quality over the UK (Heal et al., 2013; Vieno et al., 2014; Vieno et al., 2016; Nemitz et al., 2020), including provision of evidence on air quality to the UK government (HPA, 2012; AQEG, 2013; AQEG, 2017; AQEG, 2021). Details of the model system have been described previously in our companion HIA study (Macintyre et al., 2023) and in the above references, and are also detailed in the Supplementary Material, so only a brief description of the model runs used in this study is provided here.

Anthropogenic emissions of NO_x, NH_3, SO_2, CO, NMVOC (non-methane VOC), $PM_{2.5}$ and PM_{CO} (coarse particulate matter, aerodynamic diameter >2.5 μm and <10 μm) for the 2018 simulation for the UK were taken from the National Atmospheric Emissions Inventory (naei.beis.gov.uk). UK emissions for model simulations of atmospheric composition in 2030, 2040 and 2050 use the ‘business as usual (BAU)’, also referred to as ‘baseline’ scenarios developed for Defra (ApSimon et al., 2019; Defra, 2022; Oxley et al., 2023) which reflect assumed trends for anthropogenic emissions under existing interventions and policies relating to air quality, including adjustments to include the projected impact of recent policy not included in the 2018 NAEI projections. Emissions projections for the rest of the European domain use official anthropogenic EMEP emissions fields for the corresponding years (www.ceip.at). Model simulations of future air quality use 2018 meteorology in order to

[1] EMEP4UK is a nested version, focused on the British Isles (Figure 1a), of the CLRTAP EMEP MSC-W model described in Simpson et al. (2012), with updates as specified in annual reports (emep.int/mscw).

evaluate the impact of changes in anthropogenic emissions on changes in health burden. Biogenic VOC (BVOC) emissions are included in the model and are dependent on the meteorological year (here held constant to evaluate changes in anthropogenic emissions), and thus will be the same in simulations using the same meteorological year. Additionally, a sensitivity experiment was run for future years using 2003 meteorology, as an example of a hot year in the UK for comparison. The CH_4 concentration and the O_3 concentrations at the boundary of the extended European domain were fixed at 2018 values.

2.2 Exposure metric and population-weighting

The daily maximum of the 8-hour running mean O_3 concentration was calculated from gridded hourly model output for each of the simulation years at the native grid resolution, to match the O_3 exposure metric used in the epidemiology (see equations in Section 2.3). This O_3 exposure metric was population-weighted to create regional estimates for the nine regions in England, and for Scotland, Wales and Northern Ireland (Fig. 1b) using 100 m gridded residential population information for England, Scotland and Wales (National Population Database, 2020), and at 1 km for Northern Ireland (Reis et al., 2017).

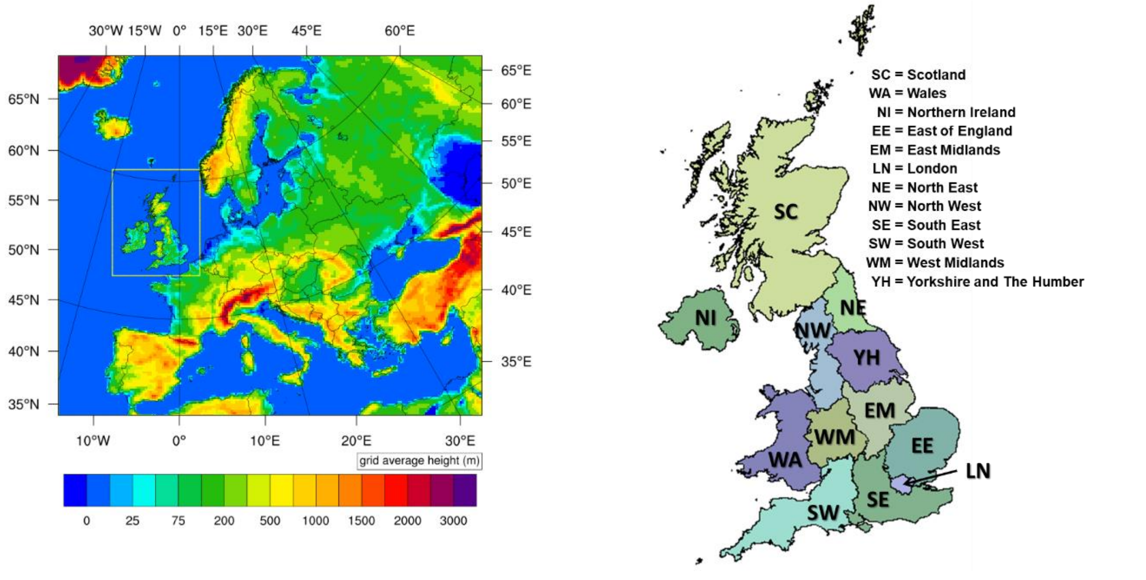


Figure 1. (a) The EMEP4UK model domain showing the 3 km x 3 km resolution British Isles domain (yellow box) nested within an extended Europe domain at 27 km x 27 km (plotted variable is altitude). (b) Regions used in the health impact assessment analysis.

2.3 Health impact calculations

The hospital admissions associated with short-term exposure to O_3 (H_{aq}) were estimated as follows: $H_{aq} = H_T \times AF$, where H_T is the total emergency respiratory hospital admissions (all-ages) in the region, and AF is attributable fraction of the health outcome in each region associated with exposure to O_3. AF is calculated based on the percent increase in emergency respiratory hospital admissions per 10 µg m^{-3} O_3 reported from meta-analyses of time-series studies. We used an exposure-response coefficient corresponding to an increase in emergency respiratory hospital admissions (all-ages) of 0.75% (95% CI: 0.30%,1.2%) per 10 µg m^{-3} daily maximum 8-hour running mean O_3 with no threshold for effects (COMEAP, 2015; COMEAP, 2022).

Daily emergency hospital admissions for respiratory causes (all-ages) were obtained from ONS for the nine regions in England (extracted via UKHSA DataLake), from Public Health Scotland for Scotland, and provided by Northern Ireland Statistical and Research Agency (NISRA) for Northern Ireland for 2018. For Wales, daily data was not available so annual totals for respiratory causes and emergency admission type were obtained from Patient Episode Database for Wales (PEDW, year 2018/19 dhcw.nhs.wales), and distributed across days in the year using the daily cycle of the data for England. Annual totals are shown in Supplementary Table S1.

To account for future changes in population, the HIA results were normalised with mid-year population estimates for 2018 for each region, and then scaled for future years using population projections. ONS produce a range of population projections for the UK based on assumptions about future fertility, mortality, and migration trends (based on long-term demographic trends), with different scenarios ('variant' projections) produced based on high/low fertility, migration, and mortality. We used the 'principal' population projections representing a middle estimate for future population demographics (interim 2020-based, (ONS, 2022)[2]). Projections are available for Scotland, Wales, Northern Ireland, and England as a whole, so regional totals in England are scaled uniformly. Population totals are shown in Supplementary Table S2.

3 Results

3.1 Impact of future emissions on O_3 and associated hospital admissions

Across the UK, population-weighted O_3 is generally highest in the spring and early summer (April to June) and lower in winter, with some very low values occurring in colder months (November to February, Fig. 2a). This seasonal cycle is consistent with surface observations

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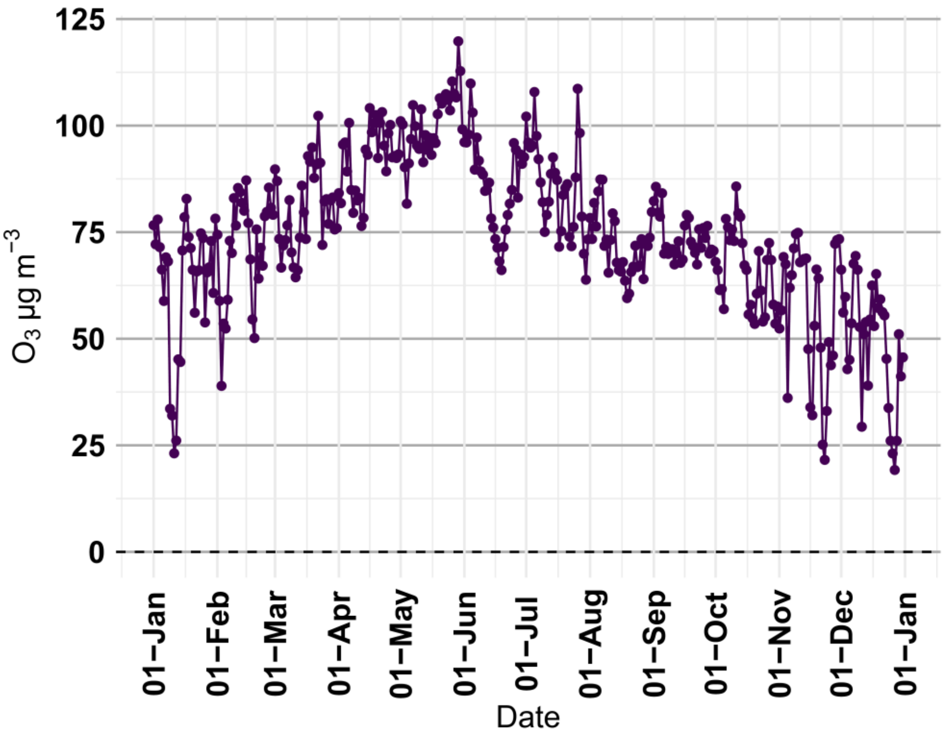
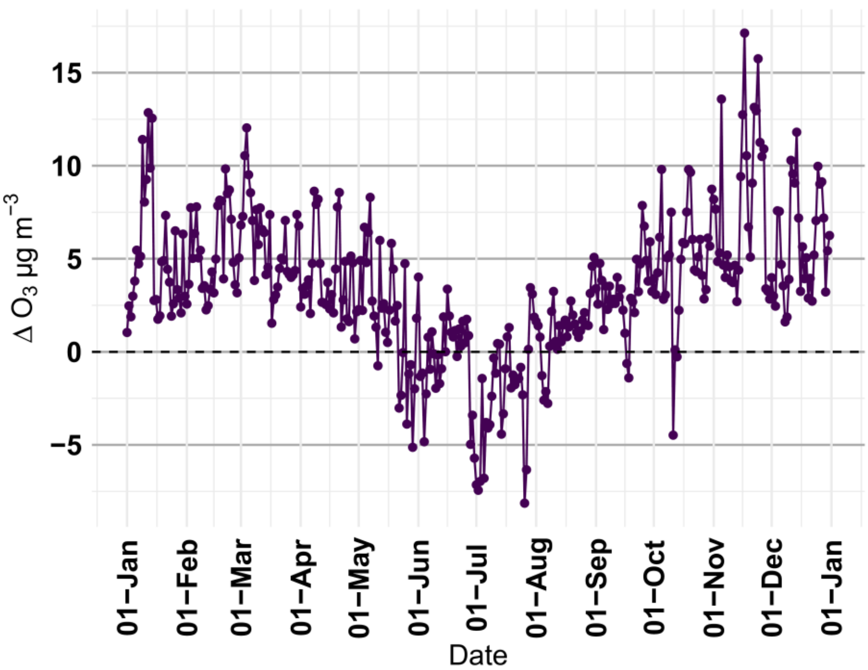
[2] The use of the term "interim" in the 2020 release reflects the interval between the 2020-based principal projection and subsequent projections, which will incorporate Census 2021 data. It also recognises uncertainties in the mid-2020 base year and in setting long-term demographic assumptions following the onset of the coronavirus (COVID-19) pandemic.

of UK O_3 (AQEG, 2021). The lowest values appear mostly in the London region, with the West Midlands, East of England, and North west also showing low excursions (Fig. 3). Comparing the impact of 2050 emissions against 2018 emissions shows an overall increase in O_3 (Fig. 2b,c). The increases in O_3 under 2050 emissions are in areas where O_3 is lowest with 2018 emissions, thus the~~with~~ greatest increases in O_3 2050 are in those regions showing lower values in 2018 (Fig. 3). In winter months (November to March) the O_3 concentrations increase in response to emission changes across all regions, whereas for warmer months the changes are mixed, with some increases and some decreases (Fig 2b).

(a) 2018 O_3 exposure (population-weighted value for the UK as a whole)



(b) Absolute difference in 2050 O_3 exposure with respect to 2018 O_3 exposure (i.e., positive indicates higher O_3 with~~in~~ 2050 emissions as compared with 2018)



(c) Relative difference in 2050 O_3 exposure with respect to 2018 O_3 exposure

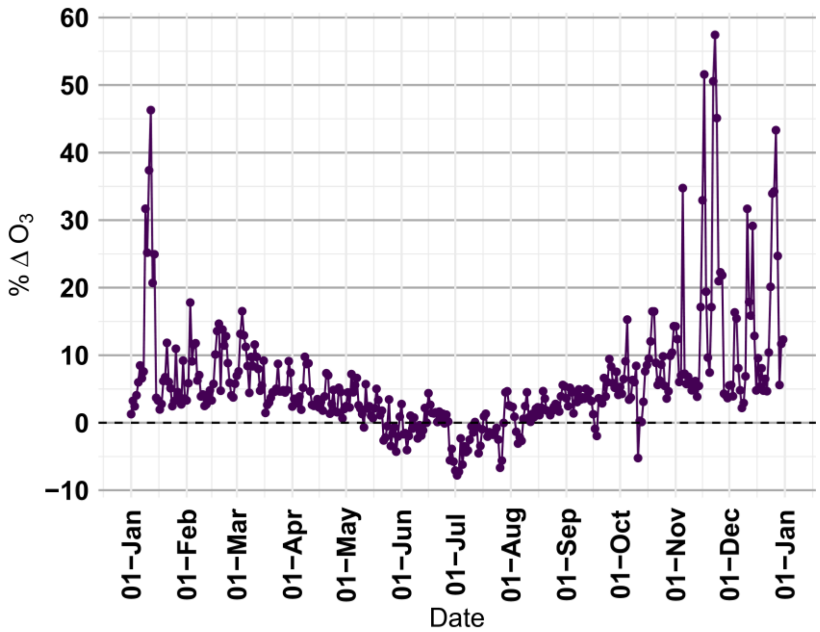
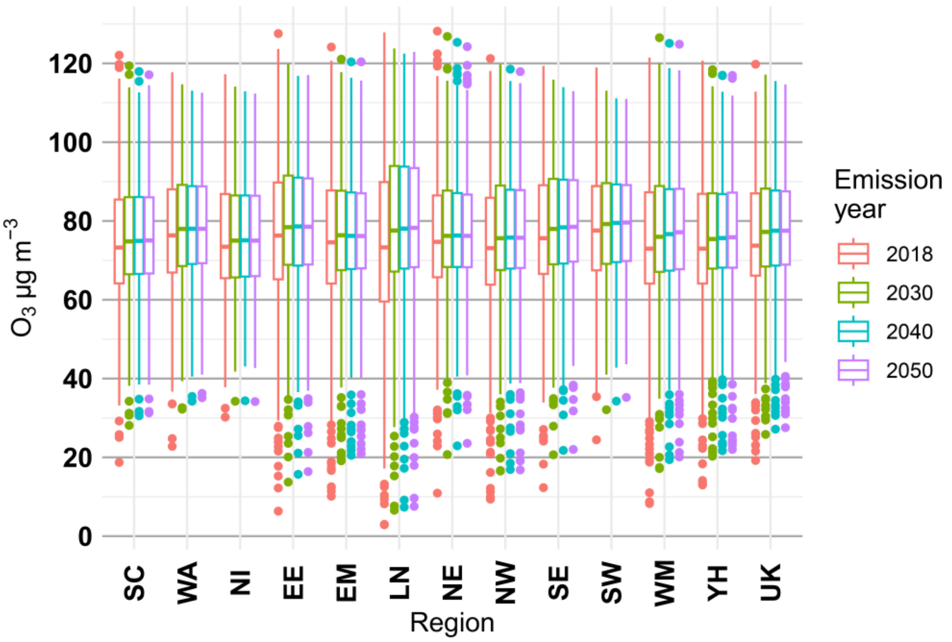


Figure 2: (a) ~~Regional~~ UK population-weighted O_3 exposure (daily maximum 8-hour running mean) ~~by country and regions in England~~ in 2018; (b) Difference in 2050 compared to 2018; (c) percent difference in 2050 compared to 2018. ~~(See Fig. 1b for region codes.)~~

Examining the overall annual distribution of O_3 exposure ~~for each of the~~by region~~s~~ and emission years (Fig. 3) reveals small increases in median daily O_3 exposure, of around 2 – 4 µg m^{-3}, and up to 6 µg m^{-3} in London, with some individual days showing >20 µg m^{-3} increase (Fig. 3b).~~,~~ There is ~~with~~ a narrowing of the range of exposures by 2050 (Fig. 3a), as high values in summer are slightly reduced, and low values in winter months are increased

(Fig. 2b). While the range of exposure narrows in future, the range in the difference compared to 2018 appears to expand (Fig. 3b) indicating less variations in the daily concentrations; this is due to the decreases in O_3 occurring where there is existing higher O_3 (so peak O_3 values in summer are reduced), and the increases in O_3 being where low O_3 values currently occur, particularly so for the instances of very low O_3 values. As the increases in O_3 (due to less suppression by NO) occur on background values that are already small, this leads to very high apparent changes when expressed as a percentage (Fig. 3c).

(a) O_3 exposure



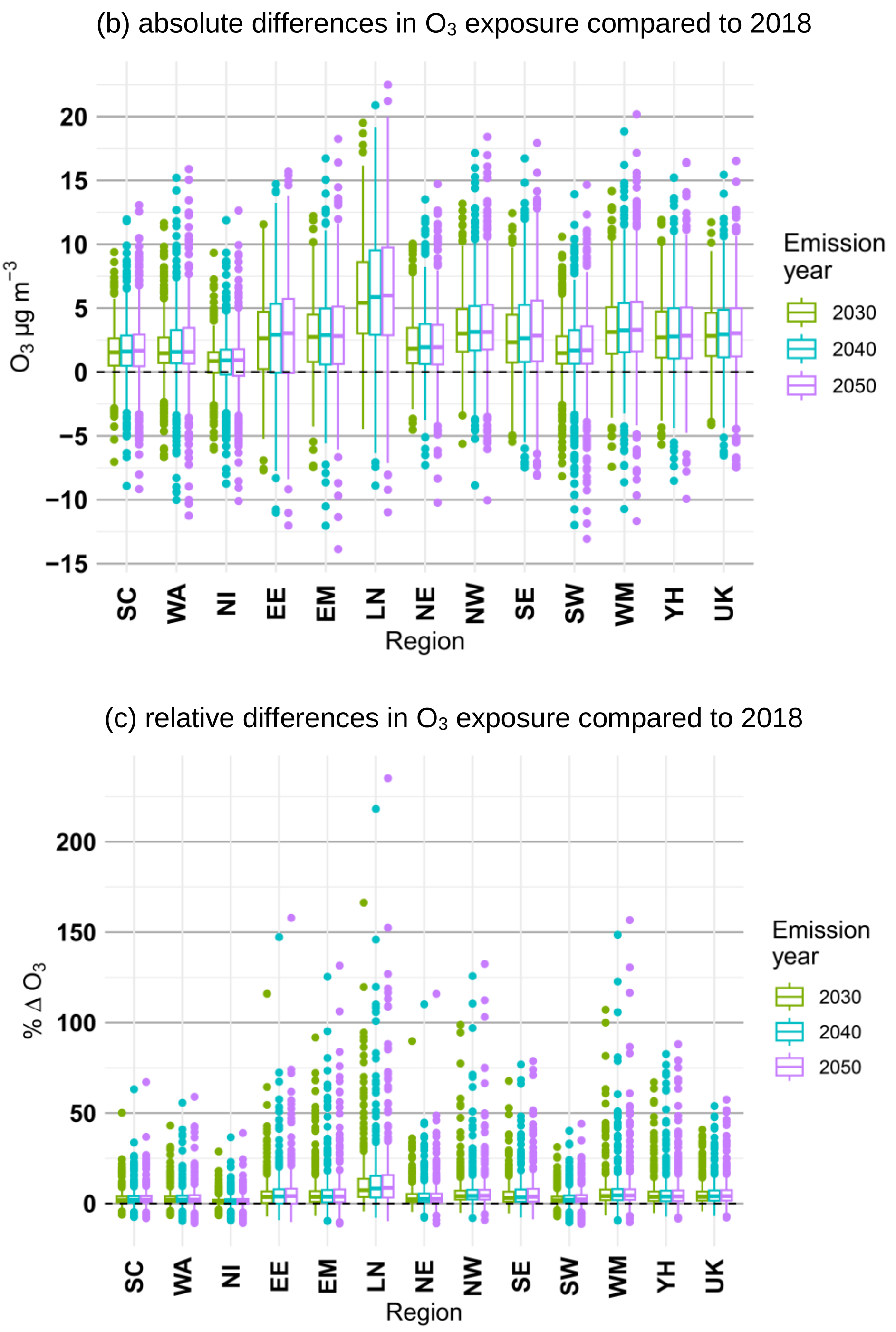


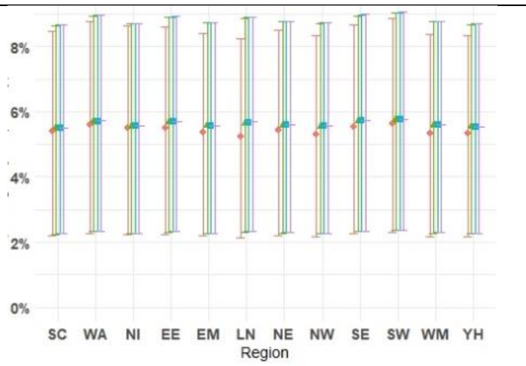
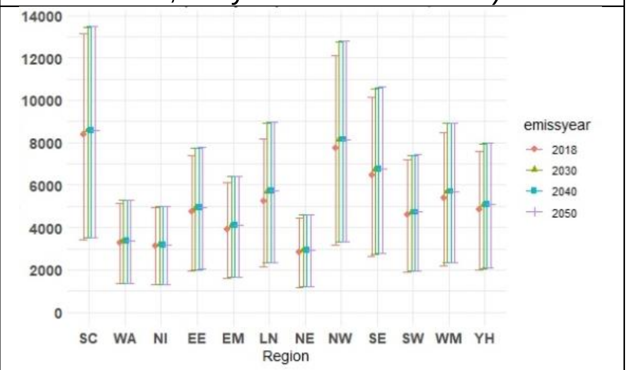
Figure 3: Boxplots* of daily population-weighted O_3 exposure (based on daily max 8-hour running mean) across all regions and different emission years (meteorology is for 2018 in all simulations). (a) Daily absolute values; (b) daily absolute differences compared to 2018; (c) percent differences compared to 2018. (See Fig. 1b for region codes; EN = England, UK= Whole UK population-weighted value.)

*Whiskers extend to the largest value or no further than 1.5 * IQR. Outliers beyond this range are plotted individually.

The annual mean daily ~~attributable~~ fraction of emergency respiratory hospital admissions attributable to short-term effects of O_3 exposure was 5.4% in 2018 (with mean ranging from

5.25% in London to 5.65% in the South West), rising slightly to 5.6% by 2050 (range 5.52% in Scotland to 5.78% in the South West) (Fig. 4). The annual cycle of the attributable fractions is in Supplementary Fig. S1. While attributable fractions are driven by air pollution concentrations, the absolute totals of attributable hospital admissions are also influenced by baseline admission rates and population in each region. Northern Ireland, Scotland, and the North West have higher baseline admission rates, and the South East, London and the North West have higher populations, leading to the highest overall attributable hospital admissions being estimated in Scotland, North West and South East (Fig. 4). Given that the changes in future O_3 concentration are comparatively small, the differences in annual attributable fractions and hospital admissions in future years compared with 2018 shown in Fig. 4 are likewise relatively modest. (The values of attributable fractions and hospital admissions using the confidence interval range on the exposure response function are shown in Supplementary Fig. S2.)

Our analysis suggests that for 2018 the annual total of daily emergency hospital admissions for respiratory causes across the UK associated with short-term exposure to O_3 was 60,488 (95% CI: 24,673 – 94,927), and that this will be higher by 4.2%, 4.5%, and 4.6% in 2030, 2040 and 2050 respectively (relative to 2018) reaching 63,289 (95% CI: 25,827 – 99,278) with 2050 emissions, an increase of 2,801 (1,154 – 4,352) (Table 1). The overall estimated associated hospital admissions in each region are also influenced by baseline admission rates and population size in each region, with the North West, London and South East having larger populations, and Scotland having larger admission rates.

<table>
<tr><td></td><td>Attributable fraction</td><td>Total hospital admissions (annual total top row, daily values bottom row)</td></tr>
<tr><td>Mean value, with 95% CI based on the exposure-response coefficient</td><td>
</td><td>
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Annual distribution (boxplot) of daily values

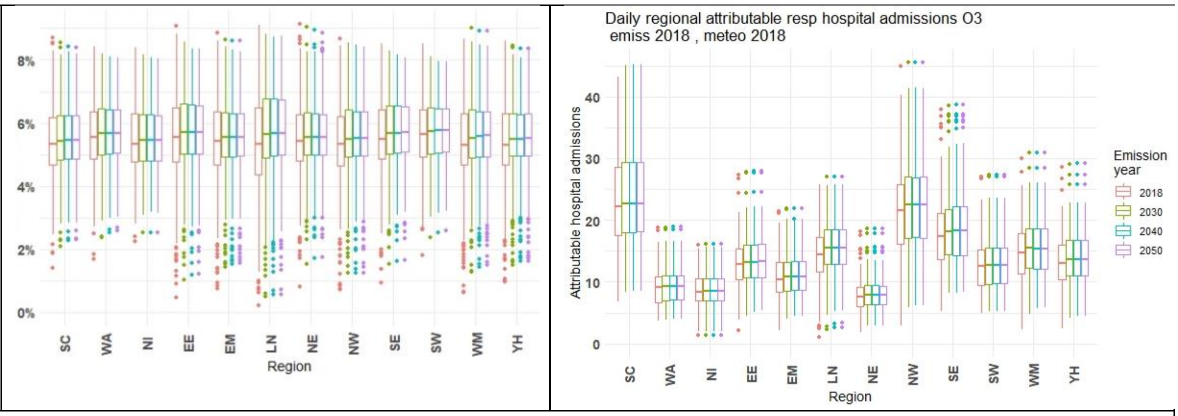

Figure 4: Regional attributable fraction (left column) and annual total respiratory emergency hospital admissions (right column) associated with short-term exposure to O_3. The top row shows annual totals, with error bars showing the 90% confidence interval on the exposure-response coefficient. The bottom row boxplots show the detail of the annual distribution of the daily values of attributable fraction and daily attributable admissions for the central estimates of future populations. (See Fig. 1b for region codes.)

Table 1: Annual total of daily emergency respiratory admissions associated with short-term O_3 exposure (95% confidence interval on the exposure response coefficient).

Region	2018	2030	2040	2050
Scotland	8366 (3412 – 13131)	8576 (3499 – 13456)	8587 (3503 – 13474)	8589 (3504 – 13477)
Wales	3277 (1337 – 5140)	3360 (1371 – 5269)	3366 (1374 – 5278)	3368 (1374 – 5282)
Northern Ireland	3148 (1284 – 4938)	3181 (1298 – 4990)	3181 (1298 – 4990)	3180 (1298 – 4989)
East of England	4724 (1927 – 7413)	4922 (2009 – 7720)	4945 (2018 – 7758)	4955 (2022 – 7771)
East Midlands	3903 (1592 – 6126)	4080 (1665 – 6400)	4093 (1670 – 6421)	4096 (1671 – 6425)
London	5222 (2130 – 8197)	5680 (2319 – 8907)	5716 (2333 – 8963)	5725 (2337 – 8978)
North East	2826 (1153 – 4435)	2921 (1192 – 4583)	2929 (1195 – 4595)	2930 (1195 – 4596)
North West	7723 (3149 – 12123)	8127 (3316 – 12750)	8150 (3326 – 12786)	8152 (3326 – 12790)
South East	6466 (2638 – 10146)	6723 (2744 – 10544)	6759 (2759 – 10601)	6778 (2766 – 10630)
South West	4605 (1879 – 7224)	4716 (1925 – 7395)	4728 (1930 – 7414)	4735 (1933 – 7425)
West Midlands	5388 (2197 – 8457)	5673 (2315 – 8900)	5695 (2324 – 8934)	5700 (2326 – 8942)
Yorkshire & Humber	4839 (1974 – 7596)	5065 (2066 – 7946)	5079 (2072 – 7970)	5082 (2073 – 7974)
UK Total	60,488 (24,673 – 94,927)	63,024 (25,719 – 98,863)	63,228 (25,802 – 98,863)	63,289 (258270 – 99,278)

When accounting for changes in population size in the future (scaled based on the population changes in Table S2), the annual total of daily hospital admissions for the UK in 2050 rises from 63,289 to 67,566 (+4,277) (Table 2), meaning that the increases from the 2018 baseline of 4.2%, 4.5%, and 4.6% in 2030, 2040 and 2050 respectively (due to O_3 changes only), are now increased to 8.3%, 10.3% and 11.7% respectively (due to both O_3 and population changes) (full breakdown of results in Table S3). This total hides some regional variation, as although overall UK population increases by 2050, the population in Scotland is expected to rise in 2030, but then fall thereafter and by 2050 is 1.9% lower than its 2018 value (Table S2). The increase in hospital admissions in Scotland (due to O_3 only with no population change) of 222 by 2050, becomes an increase of 55 hospital admissions over the same period when accounting for the shrinking population. In Northern Ireland, population is projected to increase in 2030 and 2040, but to reduce in 2050 closer to 2030 levels (Table S2). In all other regions, population changes increase the health burden, with the greatest combined (O_3 and population) changes in hospital admissions by 2050 being in London (+19.1%), the West Midlands (+14.9%), and North West (+14.7%) with respect to the 2018 baseline.

Table 2: Respiratory emergency hospital admissions associated with O_3 for future years, with and without projected population size changes.

Region	2018	2030		2040		2050	
		Base	With population change	Base	With population change	Base	With population change
England	45,698	47,907	50,101	48,094	51,348	48,153	52,319
Scotland	8,366	8,576	8,644	8,587	8,570	8,589	8,422
Wales	3,277	3,360	3,480	3,366	3,528	3,368	3,556
Northern Ireland	3,148	3,181	3,269	3,181	3,275	3,180	3,270
UK total	**60,488**	**63,024**	**65,495**	**63,228**	**66,712**	**63,289**	**67,566**

3.2 Impact of inter-annual variation in meteorology on O_3 and hospital admissions

The future emission years were also run with 2003 meteorology as another historical ‘hot’ year for the UK as a sensitivity study for the impact of interannual meteorology on O_3 concentrations. While the annual means of future population-weighted daily maximum 8-hour mean concentrations are very similar when using either 2018 or 2003 meteorology (less than 1 μg m^{-3} difference on a UK mean of all such values of 78 μg m^{-3}), there are differences in the annual cycles of the daily concentrations, as illustrated in Fig. 5 for 2050 emissions. Overall, the annual cycles both show higher concentrations in April and May (as expected), with some peaks during heatwaves and anticyclonic airflows; for example the effects on simulated O_3 of using the meteorology associated with the European heatwave in early August 2003[3] are clearly visible (Fig. 5b). There is also a particular O_3 peak associated with the 16 April 2003 meteorology, coinciding with an unusually hot day for that time of year (26.7°C was recorded in Northolt, Greater London) with clear skies and south easterly air flow, with population-weighted daily maximum 8-hour mean O_3 exceeding 120 µg m^{-3} in West Midlands, North West, East Midlands, and Yorkshire and The Humber (Supplementary Fig. S4). May 2018 was the sunniest May on record, leading to high O_3 values when using that month's meteorology, followed by a decline in O_3 in early June (6th to 19th) related to more variable weather and low-pressure systems passing over the UK (Weather, 2018a; Weather, 2018b) (Fig. 5a). A transient decrease in O_3 is also apparent in the actual measured ozone in June 2018 (Supplementary Fig. S54).

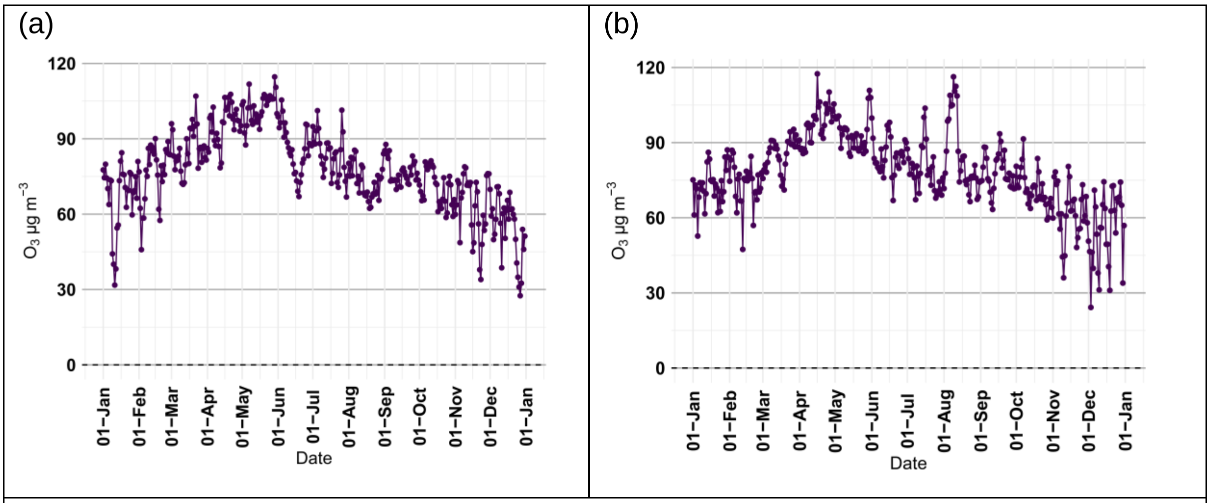

Figure 5: Population-weighted ~~regional~~ UK O_3 concentrations for 2050 emissions using (a) 2018 meteorology and (b) 2003 meteorology. ~~(See Fig. 1b for region codes.)~~

Overall totals of annual associated hospital admissions are less than 1% different when using 2018 vs 2003 meteorology (−0.46% in Scotland to 0.91% in South West for 2030 emissions, and 0.07% in Northern Ireland to 0.80% in South West for 2050 emissions). While the annual total changes are small between the two meteorological years (with the same emissions), short-term peaks and episodes of higher O_3 (and associated hospital admissions) are transiently influenced by the difference in weather patterns.

[3] "The Heatwave of 2003", Met Office, https://www.metoffice.gov.uk/weather/learn-about/weather/case-studies/heatwave. Also Vieno M, Dore AJ, Stevenson DS, Doherty R, Heal MR, Reis S, et al. Modelling surface ozone during the 2003 heat-wave in the UK. Atmos. Chem. Phys. 2010; 10: 7963-7978. 10.5194/acp-10-7963-2010.

3.3 Impact of emissions changes on hot days

As periods of hot weather and O_3 episodes often overlap, we also present the impact of emissions changes on respiratory hospital admissions with the analysis restricted to days with higher temperatures. To define a ‘hot day’, studies typically consider different percentiles of the annual distribution of daily mean temperatures, and we selected here the 95th percentile to define a ‘hot day’ (as also used by Pattenden et al. (2010)). Taking the Central England Temperature (CET) as an indicator for UK temperature (Parker et al., 1992), gives 18.4°C as the 95th percentile of the annual distribution of daily mean temperature (with a total of 34 days above this temperature in 2018). During summer when O_3 concentrations are generally higher, the impact of emission reductions is to lower some of the peak O_3 values (Fig. 2b), and results in a reduction in respiratory hospital admissions associated with O_3 exposure for these days (Table 3). This would indicate that while the overall annual total of respiratory emergency hospital admissions across the year increases in response to anthropogenic emissions changes (Fig. 4, Table 1), those admissions occurring on the hottest days (when O_3 concentrations are highest) are actually reduced, demonstrating a benefit of anthropogenic emission controls on hot days.

Table 3: Health impact assessment restricted to hot days (>95th percentile of annual daily mean CET distribution) at the UK level. (The total number of days in consideration here is 34, to consider when comparing to annual totals. Population changes are not assumed here).

Emission year	2018	2030	2040	2050
O_3 exposure (µg m^{-3})	86.1	85.1	84.2	84.0
Mean attributable fraction	6.2%	6.2%	6.1%	6.1%
Total attributable hospital admissions over the 34-days (change from 2018)	4958	4904 (−54)	4851 (−107)	4841 (−116)
Mean daily attributable hospital admissions (change from 2018)	146	144 (−1.6)	143 (−3.1)	142 (−3.4)

4 Discussion

Our results suggest that current policies driving anthropogenic emission changes will lead to an increase in future UK O_3 concentrations and hence in associated respiratory emergency hospital admissions; for example, by 4.6% in 2050, an increase of 2,801 admissions across the UK from 2018 assuming no population changes. Hospital admissions increase by 11.7% by 2050 when population projections are also included (Table S3). ~~Our p~~Previous work has shown that in future there are likely to be improvements in UK air quality through reductions in $PM_{2.5}$ and NO_2, and a reduction in the mortality burden associated with long-term exposure (Macintyre et al., 2023), whilst here we show that there may be a concurrent, albeit smaller, increase in respiratory hospital admissions. When restricting the analysis to the hottest days of the year, future anthropogenic emission reductions result in lower O_3 concentrations on hot days, and lower associated respiratory hospital admissions. While the overall annual burden of associated respiratory hospital admissions increases (as hospital admissions are generally greater in colder months, and the effect of emission reductions is generally to increase O_3 concentrations in these months), on hot days O_3 concentrations are lower in simulations with lower anthropogenic emissions, which could be beneficial for health and care services during periods of hot weather (Table 3). The use of different meteorological years in our analysis results in only very small (<1%) differences in the annual total attributable daily hospital admissions, although there is a clear influence of meteorology on day-to-day variation in O_3 concentrations and thus in associated hospital admission (Fig. 5). It is not possible to know future meteorology exactly, but clearly there will be some potential impact of interannual variability in meteorology on future O_3 concentrations and health burdens as well. As heatwaves (commonly associated with O_3 episodes) are projected to become more frequent and intense with climate change in the UK (Murphy et al., 2018), there may be implications for response planning when periods of hot weather are forecast, as there may also be health effects from increased O_3. While the aim here is to examine the impact of changes in anthropogenic emissions, the role of natural emissions of O_3 precursors (for example BVOC emissions that increase with temperature), and hemispheric CH_4 may become more important in future as anthropogenic emissions become smaller, and as temperatures rise. BVOC emissions are included in the model and are dependent on the meteorology; though as the aim of the study is to evaluate the impacts of anthropogenic air pollutant emission changes associated with current policies, we use the same meteorological year to aid comparison. However, whilst BVOC emissions are the same for simulations using the same meteorological year, the impact of the BVOC emissions on O_3 formation will vary with the different anthropogenic VOC and NO_x emissions. Our simulations extend to 2050 which has been shown to be about the point where future temperature projections start to significantly diverge for different scenarios. It is not possible to simulate future meteorology at the high spatial and temporal resolution, and over a large enough spatial domain, required to drive BVOC emissions and other meteorologically-dependent transboundary atmospheric chemistry processes within air quality models such as the EMEP4UK model we use here. Instead, our selection of a relatively hot recent year (2018) that may become a 50-50 likely summer by 2050 (as noted in the UKCP18 headline findings report) will emulate potential representation of future conditions.

Regional variation in the impacts of emissions changes is generally modest; the largest impacts of emissions changes are observed in London, which has the greatest reduction in NO_x emissions in future (leading to enhanced O_3 through reduced titration with NO). At the daily scale, the largest changes are to the lowest O_3 concentrations which are due to high NO chemically depleting the O_3. By reducing NO emissions, the most polluted NO days are reduced in future, leading to increases in O_3 on these days (Fig. 3). This effect has also been observed during the sudden curtailment of emissions, particularly from transport, during the 'lockdown' periods associated with COVID-19 measures in 2020 (AQEG, 2020; Grange et al., 2021; Jephcote et al., 2021).

Our results here align with other studies assessing the impact of future emission changes on O_3 concentrations which find similar increases near population centres, driven by NO_x emission reductions (Heal et al., 2013; Hedegaard et al., 2013; Fenech et al., 2021). Very few studies have quantified hospital admissions associated with O_3 across the UK (with the majority examining mortality); our results for 2018 (60,488) are larger than those of Heal et al. (2013, who reported 30,700 respiratory hospital admissions associated with O_3 exposure in 2003, though our study has higher baseline admission rates, different background emission scenarios and atmospheric model resolution, and a slightly larger ERF coefficient; our study also has higher O_3 levels, with averages in the range 70-80 µg m^{-3} compared with 60-70 µg m^{-3}. Stedman et al. (1997) estimate about 6% of respiratory emergency hospital admissions are attributable to O_3, which compares with the figures in our study, noting this is strongly affected by use or not of an O_3 concentration threshold, where health effects are assumed to be attributed to the exposure only above a given threshold for effects, and no health impacts attributed at concentrations below this. The choice of exposure coefficient and concentration threshold for effects is important as this impacts the overall health burden estimates, which are higher when no threshold is used (e.g. Stedman 1997 found a 20-fold increase in burden estimate when using no threshold as opposed to a 50 ppbv threshold). However, when assessing the impact of O_3 changes, the impact of the change appears larger when a threshold is used; for example if a threshold for O_3 effects is assumed, health burdens are more sensitive to changes in O_3 concentrations, although total health burdens are roughly an order of magnitude lower (Stedman et al., 1997; Stedman and Kent, 2008; Heal et al., 2013; COMEAP, 2022). Recent recommendations for the UK are not to use a threshold for effects (COMEAP, 2022), which we followed in this work. There is some evidence that temperature may act as a modifier for O_3 effects, and this may be important in future if heatwaves and episodes of O_3 become more common or intense (Pattenden et al., 2010; Atkinson et al., 2012). It is not possible to predict future changes in exposure-response relationships. Changes in baseline hospital admissions may also influence overall health impact estimates.

The UK has an objective ~~target~~ value of 100 µg m^{-3} for the daily maximum 8-hour mean O_3, not to be exceeded more than 10 times a year as set out in the National Air Quality Objectives as part of the Air Quality Standards Regulations 2010 (UK Government, 2010). Our study shows that, in future, the number of days exceeding the target value of 100 µg m^{-3} is projected to increase in London (from 44 in 2018 to 57 days in 2050), North West (26 in 2018 to 36 in 2050), and East England (37 in 2018 to 48 in 2050), and decreases in others (1 or 2 fewer days in Scotland, Northern Ireland and the South West), with small increases in other areas. In 2021, the WHO updated its recommended air quality guideline (AQG) levels for air pollutants, including O_3, with a new guideline of 100 µg m^{-3} for 8-hour mean O_3 (reduced from 120 µg m^{-3}) not to be exceeded 3-4 times per year (World Health Organization, 2021). Our analysis shows it may be harder for the UK to meet its objective and the new WHO AQG values in future. We note however that the data presented above are based on population-weighted mean values for the region or country and that changes in daily maximum 8-hour mean O_3 at specific locations will vary.

At global scale, surface O_3 is expected to decrease slightly in future in low-NO_x regimes, due to increased humidity leading to greater O_3 destruction, whereas in polluted regions, model studies suggest an increase in O_3 (Szopa et al., 2021), with the influence of climate changes on air quality being dependent on the particular emissions scenarios (Fiore et al., 2015). Here we use fixed CH_4 and O_3 boundary conditions, and while wider hemispheric trends in ozone from global trends in emissions of CH_4 will impact on UK concentrations, the direction of change is unclear (AQEG, 2021).

~~The results here and from our previous work~~A previous study using the same modelling as here ~~(Macintyre et al., 2023)~~ showed that ~~while~~ reducing emissions generally improves air pollution ($PM_{2.5}$ and NO_2) and the associated ~~health~~ mortality burdens by around a third by 2050 (Macintyre et al., 2023), though as no threshold for effects is assumed, health impacts still remain. Here however we show that ~~,~~ there can be some worsening effects related to secondary pollutants such as O_3~~.~~, though for hot days (that often coincide with higher O_3) these may be reduced. While the overall impacts of emission reductions are beneficial for health (Macintyre et al., 2023) (acknowledging that it is challenging to directly compare mortality burdens and daily hospital admissions), we find that there may be increased effects on emergency respiratory hospital admissions~~ particularly during air pollution episodes~~, which is important for health system response planning when such episodes are forecast. Our analysis in this study and a previous study (Macintyre et al., 2023)~~both studies~~ provides quantitative estimates of the potential effects to health of current policy commitments. ~~Although overall health burden is simulated to decrease in future, because there are no thresholds for effects then health impacts associated with exposure to air pollution will remain in future.~~ The methods here use a simple burden calculation to estimate health effects, though the specific impacts will depend on the speed and scale of emission reductions. Our results indicate the likely impacts to health if emissions likely to be reached at a particular indicative year in the future were to be met and sustained. While we examine the impact of population size changes, potential changes in baseline hospital admission rates are another source of uncertainty for future projections. An important point is that the UK population is ageing; as the likelihood of living with chronic disease and susceptibility to illness in general tends to increases with age, there may be increases in the baseline rates of many health effects associated with air pollution.~~,~~ Future baseline rates of hospital admissions ~~though these~~ are challenging to predict in any robust manner as many factors influence these (e.g. circulating respiratory infections, rates of chronic respiratory disease, influenza vaccine uptake and effectiveness) and thus are not considered here, which is a limitation of the study. Another limitation is related to potential future changes in exposure-response relationships; No known methodology exists to reliably predict these as these depend on multiple factors such as the future air pollution composition, exposure estimates, and population health, thus we do not examine this in the current analysis. Finally, the effects of higher air pollution levels and air pollution episodes will undoubtedly be modified by behaviours that affect exposure and inhaled dose, such as time spent indoors and exercise undertaken.

Conclusion

The effect of future emission policy on air pollution in the UK has been modelled and a quantitative HIA to estimate impacts on emergency respiratory hospital admissions~~health~~ performed. While our previous work showed significant reductions in the mortality burden due to lower $PM_{2.5}$ and NO_2 concentrations, we estimate here that there will be a small increase in O_3 exposure in future, and hence in associated hospital admissions, as a result of lower NO in urban areas, where most people reside. The main impacts on O_3 occur on the days with lowest O_3 (i.e. highest NO_x days) in regions that are already more polluted generally, such as London. Emission control policy has clear benefits to health through reducing $PM_{2.5}$ and NO_2, though effects related to secondary pollutants such as O_3 can be complex and should be acknowledged. As other air pollutants in the UK continue to decline thanks to emission controls, it seems likely that the importance of O_3 will grow. Future work on the health impacts of air pollution episodes in summer coinciding with heat should consider potential worsening effects associated with local increasing O_3 concentrations, and may need to be considered when ~~be a priority for better~~ planning ~~o~~for public health protection during these periods.

Acknowledgements

This research was partly funded by the National Institute for Health Research (NIHR) Health Protection Research Unit (HPRU) in Environmental Change (NIHR200909), a partnership between the UK Health Security Agency and the London School of Hygiene and Tropical Medicine, University College London, and the Met Office, and the HPRU in Environmental Exposures and health, a partnership between the UK Health Security Agency and the University of Leicester. The views expressed are those of the authors and not necessarily those of the NIHR, the UK Health Security Agency, or the Department of Health and Social Care. We acknowledge health data from ONS, Public Health Scotland, Northern Ireland Statistical and Research Agency (NISRA), dhcw.nhs.wales, and population data from the National Population Database, and UK Centre for Ecology & Hydrology. The work of the UK Centre for Ecology & Hydrology was supported through funding from the Department for Environment, Food and Rural Affairs under the contract “Research & Development Support for National Air Pollution Control Strategies (ECM: 62041) 2021 to 2024” and builds upon research supported by the Natural Environment Research Council award number NE/R016429/1 as part of the UK-SCAPE programme delivering National Capability. CH is supported by a NERC fellowship (NE/R01440X/1) and acknowledges funding for the HEROIC project (216035/Z/19/Z) from the Wellcome Trust.

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Future impacts of O_3 on respiratory hospital admission in the UK from current emissions policies

Helen L. Macintyre[a,b], Christina Mitsakou[a], Massimo Vieno[c], Mathew R. Heal[d], Clare Heaviside[e], Karen S. Exley[a,f]

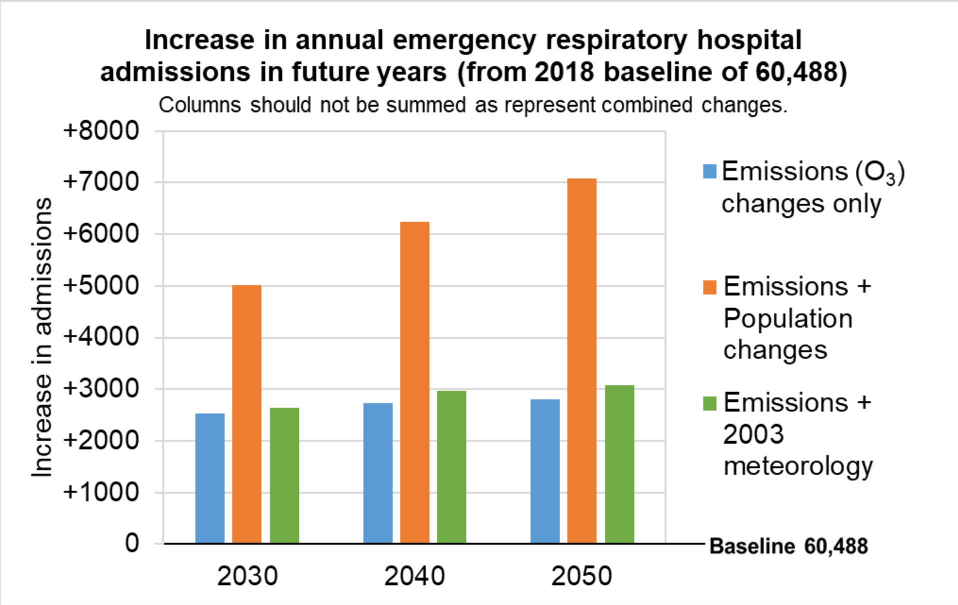
a. UK Health Security Agency, Chilton, Oxon, OX11 0RQ, UK.
b. School of Geography, Earth and Environmental Sciences, University of Birmingham, Edgbaston, B15 2TT, UK.
c. UK Centre for Ecology & Hydrology, Bush Estate, Penicuik, Midlothian, EH26 0QB, UK.
d. School of Chemistry, University of Edinburgh, Joseph Black Building, David Brewster Road, Edinburgh, EH9 3FJ, UK.
e. Institute for Environmental Design and Engineering, University College London, Central House, 14 Upper Woburn Place, London, WC1H 0NN, UK.
f. Department of Health Sciences, University of Leicester, Leicester, UK

Corresponding author: Dr Helen L Macintyre helen.macintyre@ukhsa.gov.uk
Co-author contacts: Christina Mitsakou christina.mitsakou@ukhsa.gov.uk; Massimo Vieno mvi@ceh.ac.uk; Mathew Heal m.heal@ed.ac.uk; Clare Heaviside c.heaviside@ucl.ac.uk; Karen Exley karen.exley@ukhsa.gov.uk.

Highlights

- Policies on anthropogenic emissions dominate health impacts of air pollution
- O_3 simulated at 3 km resolution for 2030, 2040 and 2050 emissions pathways
- Hospitalisations calculated with latest UK health impact assessment recommendations
- 7078 increase in respiratory hospitalisations by 2050 for current policy c.f. 2018
- Estimated increases are 2-3 times lower if population growth is not included

Graphical abstract



Keywords

Air quality; health impact assessment; emissions; respiratory hospital admissions; ozone

Abstract

Exposure to ambient ozone (O_3) is associated with impacts on human health. O_3 is a secondary pollutant whose concentrations are determined inter alia by emissions of precursors such as oxides of nitrogen (NO_x) and volatile organic compounds (VOCs), and thus future health burdens depend on policies relating to climate and air quality. While emission controls are expected to reduce levels of $PM_{2.5}$ and NO_2 and their associated mortality burdens, for secondary pollutants like O_3 the picture is less clear. Detailed assessments are necessary to provide quantitative estimates of future impacts to support decision-makers. We simulate future O_3 across the UK using a high spatial resolution atmospheric chemistry model with current UK and European policy projections for 2030, 2040 and 2050, and use UK regional population-weighting and latest recommendations on health impact assessment to quantify respiratory emergency hospital admissions associated with short-term effects of O_3. We estimate 60,488 admissions in 2018, increasing by 4.2%, 4.5% and 4.6% by 2030, 2040 and 2050 respectively (assuming a fixed population). Including future population growth, estimated emergency respiratory hospital admissions are 8.3%, 10.3% and 11.7% higher by 2030, 2040 and 2050 respectively. Increasing O_3 concentrations in future are driven by reduced nitric oxide (NO) in urban areas due to reduced emissions, with increases in O_3 mainly occurring in areas with lowest O_3 concentrations currently. Meteorology influences episodes of O_3 on a day-to-day basis,

although a sensitivity study indicates that annual totals of hospital admissions are only slightly impacted by meteorological year. While reducing emissions results in overall benefits to population health (through reduced mortality due to long-term exposure to $PM_{2.5}$ and NO_2), due to the complex chemistry, as NO emissions reduce there are associated local increases in O_3 close to population centres that may increase harms to health.

1 Introduction & background

Air pollution has detrimental impacts on human health. The main air pollutants associated with impacts on human health are $PM_{2.5}$ (particulate matter with aerodynamic diameter <2.5 μm), nitrogen dioxide (NO_2), and ozone (O_3), with long-term exposure to ambient $PM_{2.5}$ and NO_2 in the UK estimated to have an effect equivalent to 29,000–43,000 deaths annually (Mitsakou et al., 2022). Exposure to ambient O_3 is also associated with ill health, particularly respiratory effects including exacerbation of asthma symptoms. O_3 is estimated to contribute to over 5 million disability adjusted life years (DALYs) lost annually globally (Forouzanfar et al., 2015). The evidence of O_3 impacts on health is strongest for associations between short-term exposure (as quantified using the daily maximum 8-hour running mean O_3) and respiratory and cardiovascular hospital admissions, though with more uncertainty for the latter (COMEAP, 2015). Quantification of the impact on health of long-term exposure to O_3 is not recommended by the UK's Committee on the Medical Effects of air Pollutants (COMEAP) as there is currently insufficient evidence (COMEAP, 2015; Huangfu and Atkinson, 2020; COMEAP, 2022).

Ozone is not directly emitted but is a secondary pollutant formed through complex (photo)chemical reactions in the atmosphere involving NO_x (= NO + NO_2) and volatile organic compound (VOC) emissions, including of methane (CH_4). The levels of methane contribute to the longer-term (annual average) concentrations of O_3 that are also part of O_3 exposure and upon which the shorter-term variations in O_3 driven by non-methane VOCs are superimposed. Concentrations of O_3 are also affected by meteorology, which influences its build-up through photochemistry and air mass transport and dilution, and also via the rate of O_3 deposition to the surface (Royal Society, 2008). High levels of NO_x (e.g. close to emission sources) also act to limit O_3 formation, which is why reductions in NO_x emissions in urban areas, largely associated with improvements in vehicle exhaust emission standards, have led to modelled local increases in O_3 (Carnell et al., 2019).

Air pollution levels have generally improved in the UK over the past several decades, thanks to environmental and public health initiatives. Carnell et al. (2019 reported that reductions in air pollution levels between 1970 and 2010 led to a reduction in UK attributable mortality associated with exposure to $PM_{2.5}$ and NO_2 of 56% and 44% respectively; however, the same study reported an increase in O_3-attributable respiratory mortality of 17%. This reflects a different trend in response to recent air pollutant emissions changes for O_3 in urban areas, where most people live, than for urban NO_2 and $PM_{2.5}$ (AQEG, 2021). In particular, as noted above, there is potential for local increases in O_3 due to reductions in emissions of NO, with a recent report noting upwards trend in urban ozone of 5 – 9 µg m^{-3} between 2000-2019 (AQEG, 2021). A modelling study on possible UK net-zero policies showed that while longer-term (April to September) mean O_3 concentrations may reduce, short-term O_3 metrics revealed less O_3 suppression from reduced NO_x, particularly in winter (Williams et al., 2018). The effect of reduced traffic NO_x emissions increasing urban O_3 was also illustrated during the travel restrictions implemented as part of COVID-19 ‘lockdown’ restrictions (AQEG, 2020; Lee et al., 2020; Jephcote et al., 2021).

Changes in climate will also modify concentrations of surface O_3, as it will for other air pollutants (von Schneidemesser et al., 2015; Doherty et al., 2017). However, the Doherty et al. review concludes that for O_3 over Europe in the next few decades, changes in anthropogenic emissions of the VOC and NO_x precursors will have much greater influence on O_3 levels than changes in climate. For example, model simulations of impacts of climate scenarios and compatible air pollutant emissions under CMIP5 scenarios showed O_3 exposure metrics over Europe in 2050 change by 3% (±8%) to 5% (±11%) due to changes in climate, but by −24% (±10%) to −43% (±7%) due to emission changes (Colette et al., 2013). Over the UK, changes in annual mean O_3 by 2030 (compared to 2003) due to changes in precursor emissions (−3.0 to +3.5 ppbv) are larger than for a +5°C scenario (+1.0 to +1.5 ppbv) including temperature effects on biogenic VOCs, chemical reaction rates and O_3 dry deposition, with the latter being comparable to the changes in O_3 arising from normal interannual meteorological variability (Heal et al., 2013). Resulting changes in mortality burdens were 4.1% associated with O_3 changes due to +5°C, whereas they were 15.6% to 27.7% for emissions changes (depending on emission scenario), assuming no threshold for effects (Heal et al., 2013).

In a previous study we performed an up-to-date quantitative impact assessment of the mortality burdens associated with long-term effects of $PM_{2.5}$ and NO_2 in 2030, 2040 and 2050 using a high spatial resolution atmospheric chemistry and aerosol model with the latest official UK and European emissions datasets (Macintyre et al., 2023). We found that mortality burdens associated with these two pollutants were reduced by over 30% beyond 2030. As O_3 is also impacted by such emission policies, but in more complex ways as described above, we expand our analysis here to consider impacts on respiratory hospital admissions associated with short-term exposure to O_3, using a similar health impact assessment (HIA), to more fully capture future impacts to health associated with changing emissions.

2 Methods

2.1 Air pollution modelling and emissions

Simulations of UK present-day and future air quality at 3 km × 3 km spatial resolution and hourly temporal resolution were undertaken using the EMEP4UK[1] atmospheric chemistry transport model version rv4.36. The chemical model was driven by hourly meteorology from the Weather Research and Forecasting (WRF) model version 4.2.2 (wrf-model.org). The EMEP4UK-WRF modelling system has been widely used to simulate historic, present-day and potential future air quality over the UK (Heal et al., 2013; Vieno et al., 2014; Vieno et al., 2016; Nemitz et al., 2020), including provision of evidence on air quality to the UK government (HPA, 2012; AQEG, 2013; AQEG, 2017; AQEG, 2021). Details of the model system have been described previously in our companion HIA study (Macintyre et al., 2023) and in the above references, and are also detailed in the Supplementary Material, so only a brief description of the model runs used in this study is provided here.

Anthropogenic emissions of NO_x, NH_3, SO_2, CO, NMVOC (non-methane VOC), $PM_{2.5}$ and PM_{CO} (coarse particulate matter, aerodynamic diameter >2.5 μm and <10 μm) for the 2018 simulation for the UK were taken from the National Atmospheric Emissions Inventory (naei.beis.gov.uk). UK emissions for model simulations of atmospheric composition in 2030, 2040 and 2050 use the 'business as usual (BAU)', also referred to as 'baseline' scenarios developed for Defra (ApSimon et al., 2019; Defra, 2022; Oxley et al., 2023) which reflect assumed trends for anthropogenic emissions under existing interventions and policies relating to air quality, including adjustments to include the projected impact of recent policy not included in the 2018 NAEI projections. Emissions projections for the rest of the European domain use official anthropogenic EMEP emissions fields for the corresponding years (www.ceip.at). Model simulations of future air quality use 2018 meteorology in order to

[1] EMEP4UK is a nested version, focused on the British Isles (Figure 1a), of the CLRTAP EMEP MSC-W model described in Simpson et al. (2012), with updates as specified in annual reports (emep.int/mscw).

evaluate the impact of changes in anthropogenic emissions on changes in health burden. Biogenic VOC (BVOC) emissions are included in the model and are dependent on the meteorological year (here held constant to evaluate changes in anthropogenic emissions), and thus will be the same in simulations using the same meteorological year. Additionally, a sensitivity experiment was run for future years using 2003 meteorology, as an example of a hot year in the UK for comparison. The CH_4 concentration and the O_3 concentrations at the boundary of the extended European domain were fixed at 2018 values.

2.2 Exposure metric and population-weighting

The daily maximum of the 8-hour running mean O_3 concentration was calculated from gridded hourly model output for each of the simulation years at the native grid resolution, to match the O_3 exposure metric used in the epidemiology (see equations in Section 2.3). This O_3 exposure metric was population-weighted to create regional estimates for the nine regions in England, and for Scotland, Wales and Northern Ireland (Fig. 1b) using 100 m gridded residential population information for England, Scotland and Wales (National Population Database, 2020), and at 1 km for Northern Ireland (Reis et al., 2017).

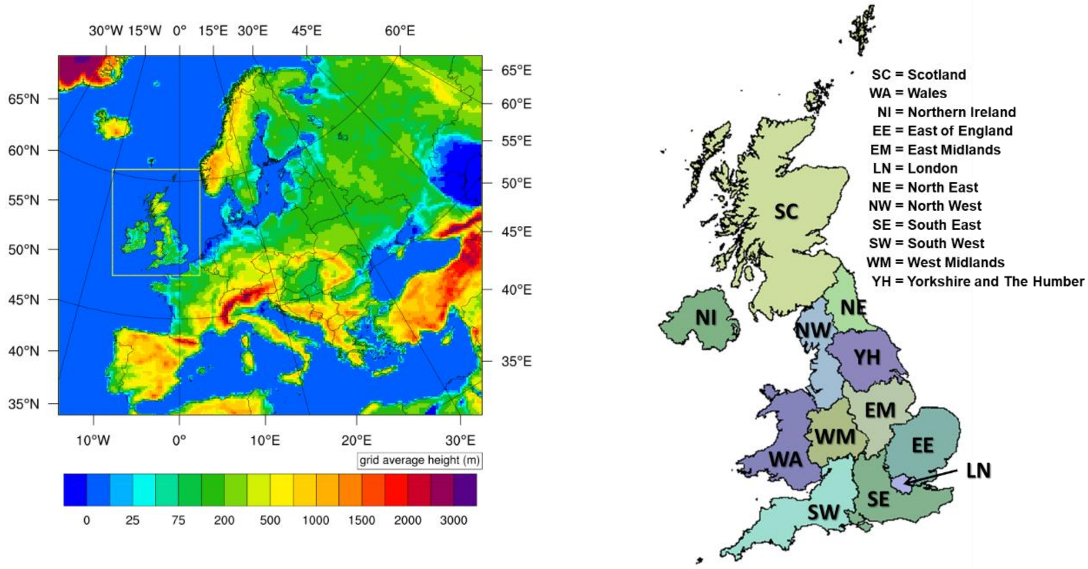


Figure 1. (a) The EMEP4UK model domain showing the 3 km x 3 km resolution British Isles domain (yellow box) nested within an extended Europe domain at 27 km x 27 km (plotted variable is altitude). (b) Regions used in the health impact assessment analysis.

2.3 Health impact calculations

The hospital admissions associated with short-term exposure to O_3 (H_{aq}) were estimated as follows: $H_{aq} = H_T \times AF$, where H_T is the total emergency respiratory hospital admissions (all-ages) in the region, and AF is attributable fraction of the health outcome in each region associated with exposure to O_3. AF is calculated based on the percent increase in emergency respiratory hospital admissions per 10 µg m^{-3} O_3 reported from meta-analyses of time-series studies. We used an exposure-response coefficient corresponding to an increase in emergency respiratory hospital admissions (all-ages) of 0.75% (95% CI: 0.30%,1.2%) per 10 µg m^{-3} daily maximum 8-hour running mean O_3 with no threshold for effects (COMEAP, 2015; COMEAP, 2022).

Daily emergency hospital admissions for respiratory causes (all-ages) were obtained from ONS for the nine regions in England (extracted via UKHSA DataLake), from Public Health Scotland for Scotland, and provided by Northern Ireland Statistical and Research Agency (NISRA) for Northern Ireland for 2018. For Wales, daily data was not available so annual totals for respiratory causes and emergency admission type were obtained from Patient Episode Database for Wales (PEDW, year 2018/19 dhcw.nhs.wales), and distributed across days in the year using the daily cycle of the data for England. Annual totals are shown in Supplementary Table S1.

To account for future changes in population, the HIA results were normalised with mid-year population estimates for 2018 for each region, and then scaled for future years using population projections. ONS produce a range of population projections for the UK based on assumptions about future fertility, mortality, and migration trends (based on long-term demographic trends), with different scenarios (‘variant’ projections) produced based on high/low fertility, migration, and mortality. We used the ‘principal’ population projections representing a middle estimate for future population demographics (interim 2020-based, (ONS, 2022)[2]). Projections are available for Scotland, Wales, Northern Ireland, and England as a whole, so regional totals in England are scaled uniformly. Population totals are shown in Supplementary Table S2.

3 Results

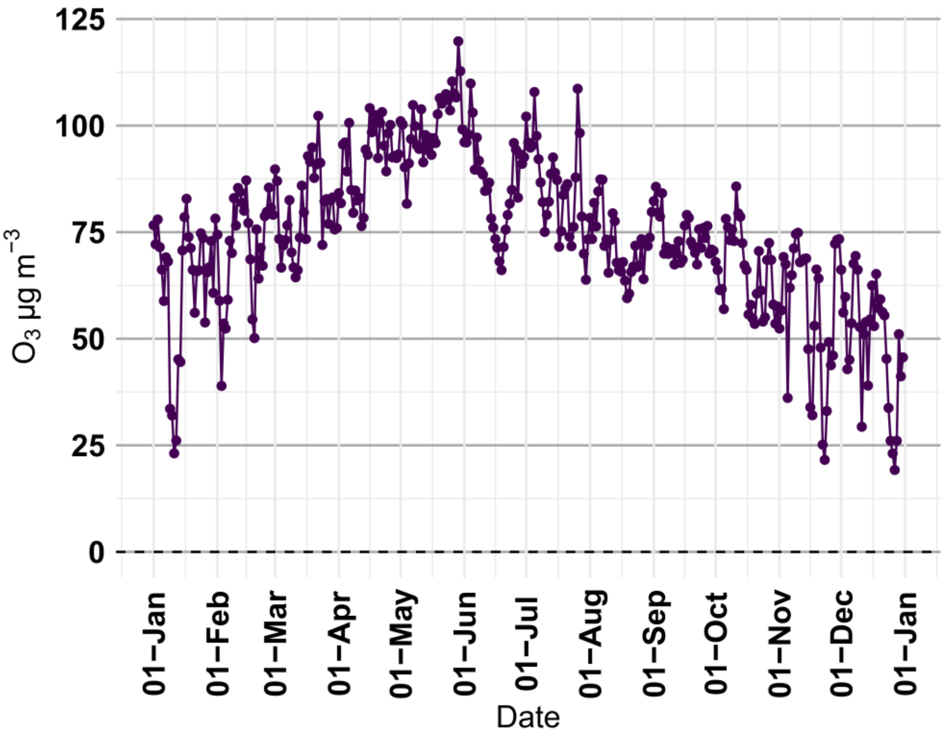
3.1 Impact of future emissions on O_3 and associated hospital admissions

Across the UK, population-weighted O_3 is generally highest in the spring and early summer (April to June) and lower in winter, with some very low values occurring in colder months (November to February, Fig. 2a). This seasonal cycle is consistent with surface observations

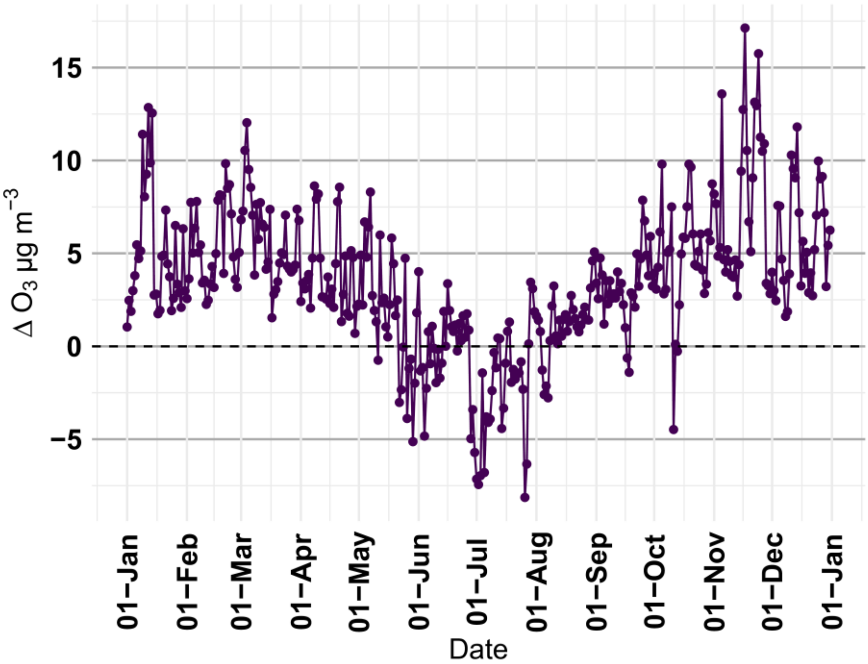
[2] The use of the term “interim” in the 2020 release reflects the interval between the 2020-based principal projection and subsequent projections, which will incorporate Census 2021 data. It also recognises uncertainties in the mid-2020 base year and in setting long-term demographic assumptions following the onset of the coronavirus (COVID-19) pandemic.

of UK O_3 (AQEG, 2021). The lowest values appear mostly in the London region, with the West Midlands, East of England, and North west also showing low excursions (Fig. 3). Comparing the impact of 2050 emissions against 2018 emissions shows an overall increase in O_3 (Fig. 2b,c). The increases in O_3 under 2050 emissions are in areas where O_3 is lowest with 2018 emissions, thus the greatest increases in O_3 2050 are in those regions showing lower values in 2018 (Fig. 3). In winter months (November to March) the O_3 concentrations increase in response to emission changes across all regions, whereas for warmer months the changes are mixed, with some increases and some decreases (Fig 2b).

(a) 2018 O_3 exposure (population-weighted value for the UK as a whole)



(b) Absolute difference in 2050 O_3 exposure with respect to 2018 O_3 exposure (i.e., positive indicates higher O_3 with 2050 emissions as compared with 2018)



(c) Relative difference in 2050 O_3 exposure with respect to 2018 O_3 exposure

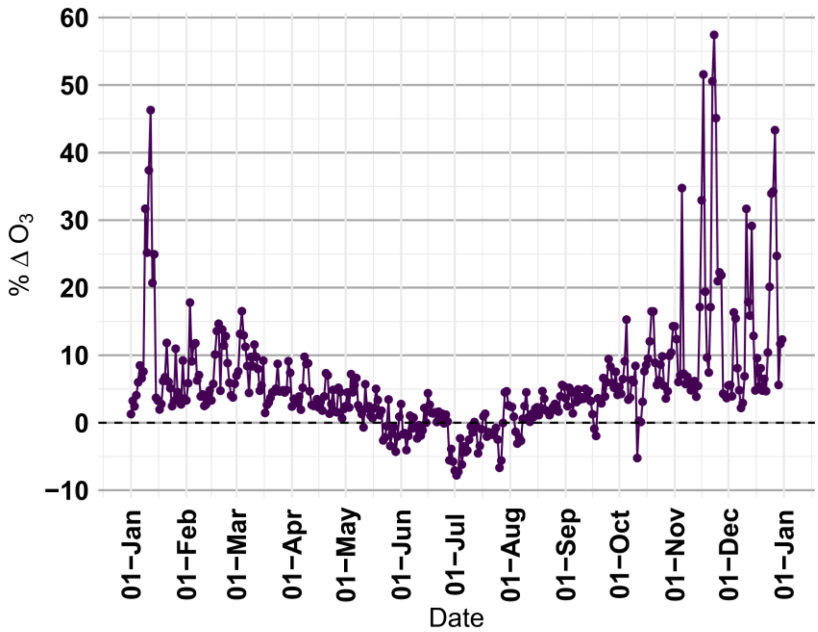
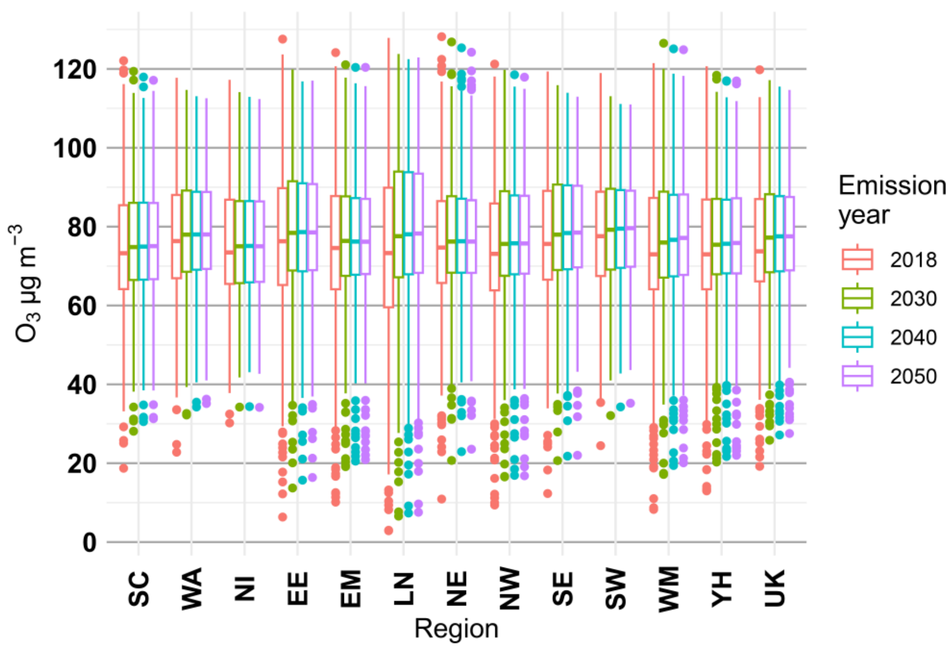


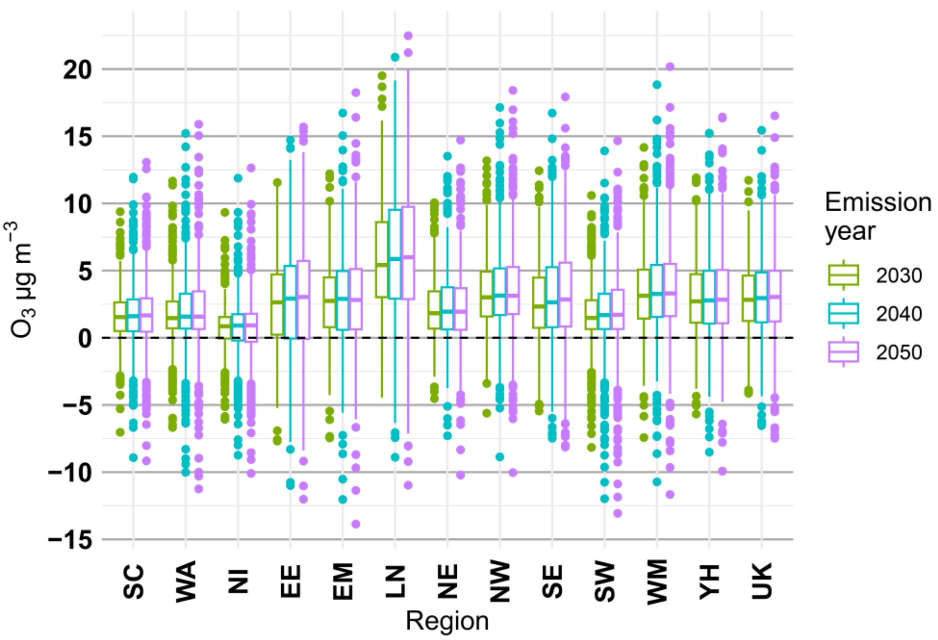
Figure 2: (a) UK population-weighted O_3 exposure (daily maximum 8-hour running mean) in 2018; (b) Difference in 2050 compared to 2018; (c) percent difference in 2050 compared to 2018.

Examining the overall annual distribution of O_3 exposure by region and emission years (Fig. 3) reveals small increases in median daily O_3 exposure, of around 2 – 4 μg m^{-3}, and up to 6 μg m^{-3} in London, with some individual days showing >20 μg m^{-3} increase (Fig. 3b). There is a narrowing of the range of exposures by 2050 (Fig. 3a), as high values in summer are slightly reduced, and low values in winter months are increased (Fig. 2b). While the range of exposure narrows in future, the range in the difference compared to 2018 appears to expand (Fig. 3b) indicating less variations in the daily concentrations; this is due to the decreases in O_3 occurring where there is existing higher O_3 (so peak O_3 values in summer are reduced), and the increases in O_3 being where low O_3 values currently occur, particularly so for the instances of very low O_3 values. As the increases in O_3 (due to less suppression by NO) occur on background values that are already small, this leads to very high apparent changes when expressed as a percentage (Fig. 3c).

(a) O_3 exposure



(b) absolute differences in O_3 exposure compared to 2018



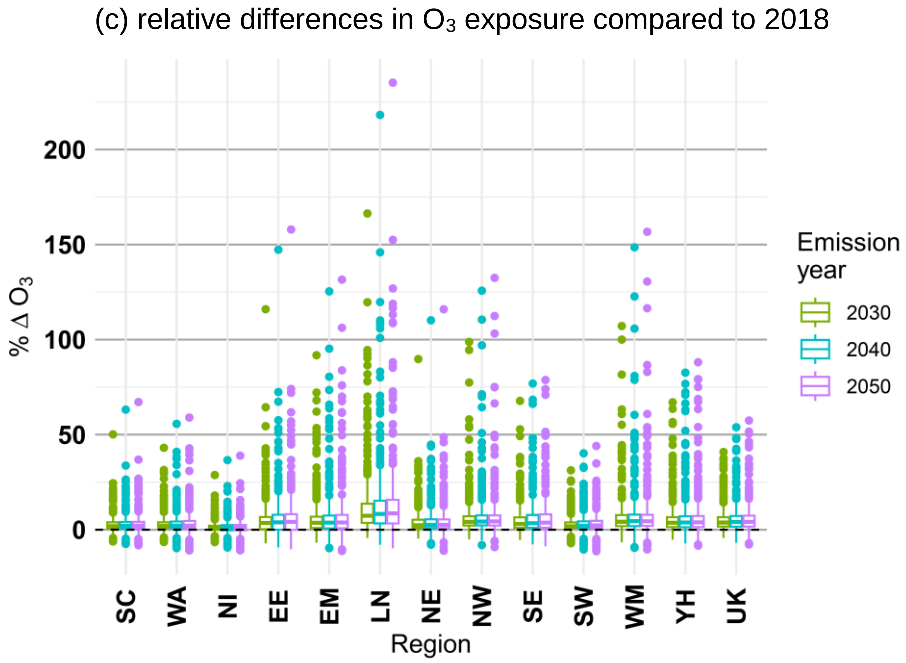


Figure 3: Boxplots* of daily population-weighted O_3 exposure (based on daily max 8-hour running mean) across all regions and different emission years (meteorology is for 2018 in all simulations). (a) Daily absolute values; (b) daily absolute differences compared to 2018; (c) percent differences compared to 2018. (See Fig. 1b for region codes; EN = England, UK= Whole UK population-weighted value.)

*Whiskers extend to the largest value or no further than 1.5 * IQR. Outliers beyond this range are plotted individually.

The annual mean daily fraction of emergency respiratory hospital admissions attributable to short-term effects of O_3 exposure was 5.4% in 2018 (with mean ranging from 5.25% in London to 5.65% in the South West), rising slightly to 5.6% by 2050 (range 5.52% in Scotland to 5.78% in the South West) (Fig. 4). The annual cycle of the attributable fractions is in Supplementary Fig. S1. While attributable fractions are driven by air pollution concentrations, the absolute totals of attributable hospital admissions are also influenced by baseline admission rates and population in each region. Northern Ireland, Scotland, and the North West have higher baseline admission rates, and the South East, London and the North West have higher populations, leading to the highest overall attributable hospital admissions being estimated in Scotland, North West and South East (Fig. 4). Given that the changes in future O_3 concentration are comparatively small, the differences in annual attributable fractions and hospital admissions in future years compared with 2018 shown in Fig. 4 are likewise relatively modest. (The values of attributable fractions and hospital admissions using the confidence interval range on the exposure response function are shown in Supplementary Fig. S2.)

Our analysis suggests that for 2018 the annual total of daily emergency hospital admissions for respiratory causes across the UK associated with short-term exposure to O_3 was 60,488 (95% CI: 24,673 – 94,927), and that this will be higher by 4.2%, 4.5%, and 4.6% in 2030, 2040 and 2050 respectively (relative to 2018) reaching 63,289 (95% CI: 25,827 – 99,278) with 2050 emissions, an increase of 2,801 (1,154 – 4,352) (Table 1). The overall estimated associated hospital admissions in each region are also influenced by baseline admission rates and population size in each region, with the North West, London and South East having larger populations, and Scotland having larger admission rates.

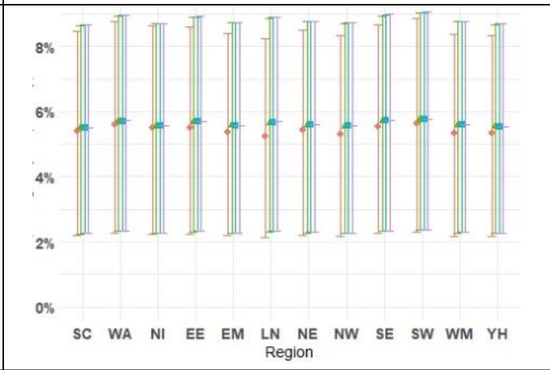
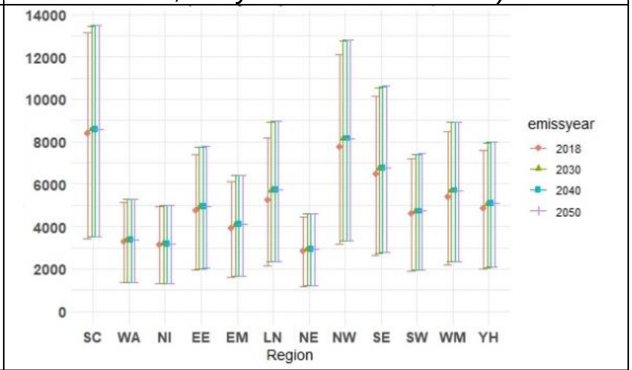
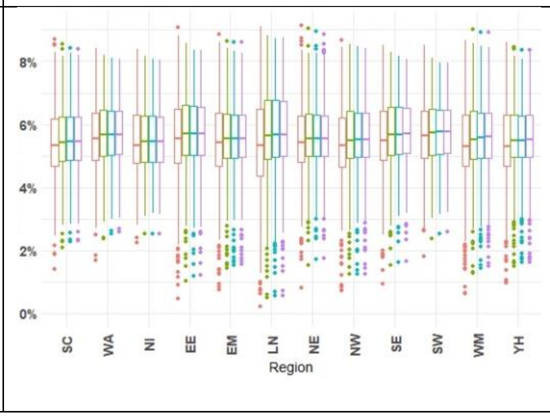
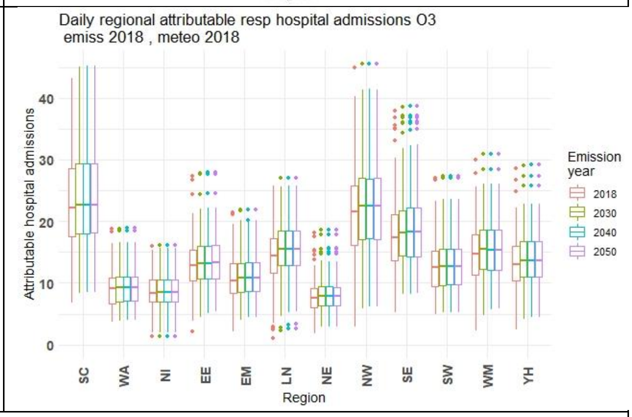
	Attributable fraction	Total hospital admissions (annual total top row, daily values bottom row)
Mean value, with 95% CI based on the exposure-response coefficient		
Annual distribution (boxplot) of daily values		

Figure 4: Regional attributable fraction (left column) and annual total respiratory emergency hospital admissions (right column) associated with short-term exposure to O_3. The top row shows annual totals, with error bars showing the 90% confidence interval on the exposure-response coefficient. The bottom row boxplots show the detail of the annual distribution of the daily values of attributable fraction and daily attributable admissions for the central estimates of future populations. (See Fig. 1b for region codes.)

Table 1: Annual total of daily emergency respiratory admissions associated with short-term O_3 exposure (95% confidence interval on the exposure response coefficient).

Region	2018	2030	2040	2050
Scotland	8366 (3412 – 13131)	8576 (3499 – 13456)	8587 (3503 – 13474)	8589 (3504 – 13477)
Wales	3277 (1337 – 5140)	3360 (1371 – 5269)	3366 (1374 – 5278)	3368 (1374 – 5282)

Northern Ireland	3148 (1284 – 4938)	3181 (1298 – 4990)	3181 (1298 – 4990)	3180 (1298 – 4989)
East of England	4724 (1927 – 7413)	4922 (2009 – 7720)	4945 (2018 – 7758)	4955 (2022 – 7771)
East Midlands	3903 (1592 – 6126)	4080 (1665 – 6400)	4093 (1670 – 6421)	4096 (1671 – 6425)
London	5222 (2130 – 8197)	5680 (2319 – 8907)	5716 (2333 – 8963)	5725 (2337 – 8978)
North East	2826 (1153 – 4435)	2921 (1192 – 4583)	2929 (1195 – 4595)	2930 (1195 – 4596)
North West	7723 (3149 – 12123)	8127 (3316 – 12750)	8150 (3326 – 12786)	8152 (3326 – 12790)
South East	6466 (2638 – 10146)	6723 (2744 – 10544)	6759 (2759 – 10601)	6778 (2766 – 10630)
South West	4605 (1879 – 7224)	4716 (1925 – 7395)	4728 (1930 – 7414)	4735 (1933 – 7425)
West Midlands	5388 (2197 – 8457)	5673 (2315 – 8900)	5695 (2324 – 8934)	5700 (2326 – 8942)
Yorkshire & Humber	4839 (1974 – 7596)	5065 (2066 – 7946)	5079 (2072 – 7970)	5082 (2073 – 7974)
UK Total	60,488 (24,673 – 94,927)	63,024 (25,719 – 98,863)	63,228 (25,802 – 98,863)	63,289 (258270 – 99,278)

When accounting for changes in population size in the future (scaled based on the population changes in Table S2), the annual total of daily hospital admissions for the UK in 2050 rises from 63,289 to 67,566 (+4,277) (Table 2), meaning that the increases from the 2018 baseline of 4.2%, 4.5%, and 4.6% in 2030, 2040 and 2050 respectively (due to O_3 changes only), are now increased to 8.3%, 10.3% and 11.7% respectively (due to both O_3 and population changes) (full breakdown of results in Table S3). This total hides some regional variation, as although overall UK population increases by 2050, the population in Scotland is expected to rise in 2030, but then fall thereafter and by 2050 is 1.9% lower than its 2018 value (Table S2). The increase in hospital admissions in Scotland (due to O_3 only with no population change) of 222 by 2050, becomes an increase of 55 hospital admissions over the same period when accounting for the shrinking population. In Northern Ireland, population is projected to increase in 2030 and 2040, but to reduce in 2050 closer to 2030 levels (Table S2). In all other regions, population changes increase the health burden, with the greatest combined (O_3 and population) changes in hospital admissions by 2050 being in London (+19.1%), the West Midlands (+14.9%), and North West (+14.7%) with respect to the 2018 baseline.

Table 2: Respiratory emergency hospital admissions associated with O_3 for future years, with and without projected population size changes.

Region	2018	2030		2040		2050	
		Base	With population change	Base	With population change	Base	With population change
England	45,698	47,907	50,101	48,094	51,348	48,153	52,319
Scotland	8,366	8,576	8,644	8,587	8,570	8,589	8,422
Wales	3,277	3,360	3,480	3,366	3,528	3,368	3,556
Northern Ireland	3,148	3,181	3,269	3,181	3,275	3,180	3,270
UK total	**60,488**	**63,024**	**65,495**	**63,228**	**66,712**	**63,289**	**67,566**

3.2 Impact of inter-annual variation in meteorology on O_3 and hospital admissions

The future emission years were also run with 2003 meteorology as another historical 'hot' year for the UK as a sensitivity study for the impact of interannual meteorology on O_3 concentrations. While the annual means of future population-weighted daily maximum 8-hour mean concentrations are very similar when using either 2018 or 2003 meteorology (less than 1 μg m^{-3} difference on a UK mean of all such values of 78 μg m^{-3}), there are differences in the annual cycles of the daily concentrations, as illustrated in Fig. 5 for 2050 emissions. Overall, the annual cycles both show higher concentrations in April and May (as expected), with some peaks during heatwaves and anticyclonic airflows; for example the effects on simulated O_3 of using the meteorology associated with the European heatwave in early August 2003[3] are clearly visible (Fig. 5b). There is also a particular O_3 peak associated with the 16 April 2003 meteorology, coinciding with an unusually hot day for that time of year (26.7°C was recorded in Northolt, Greater London) with clear skies and south easterly air flow, with population-weighted daily maximum 8-hour mean O_3 exceeding 120 μg m^{-3} in West Midlands, North West, East Midlands, and Yorkshire and The Humber (Supplementary Fig. S4). May 2018 was the sunniest May on record, leading to high O_3 values when using that month's meteorology, followed by a decline in O_3 in early June (6th to 19th) related to more variable weather and low-pressure systems passing over the UK (Weather, 2018a; Weather, 2018b) (Fig. 5a). A transient decrease in O_3 is also apparent in the actual measured ozone in June 2018 (Supplementary Fig. S5).

(a)	(b)

[3] "The Heatwave of 2003", Met Office, https://www.metoffice.gov.uk/weather/learn-about/weather/case-studies/heatwave. Also Vieno M, Dore AJ, Stevenson DS, Doherty R, Heal MR, Reis S, et al. Modelling surface ozone during the 2003 heat-wave in the UK. Atmos. Chem. Phys. 2010; 10: 7963-7978. 10.5194/acp-10-7963-2010.

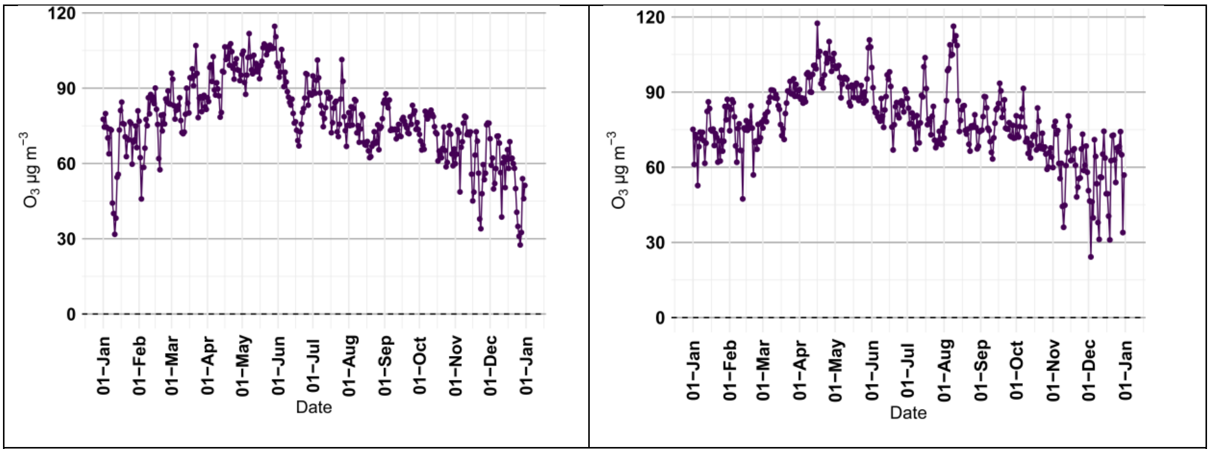


Figure 5: Population-weighted UK O_3 concentrations for 2050 emissions using (a) 2018 meteorology and (b) 2003 meteorology.

Overall totals of annual associated hospital admissions are less than 1% different when using 2018 vs 2003 meteorology (−0.46% in Scotland to 0.91% in South West for 2030 emissions, and 0.07% in Northern Ireland to 0.80% in South West for 2050 emissions). While the annual total changes are small between the two meteorological years (with the same emissions), short-term peaks and episodes of higher O_3 (and associated hospital admissions) are transiently influenced by the difference in weather patterns.

3.3 Impact of emissions changes on hot days

As periods of hot weather and O_3 episodes often overlap, we also present the impact of emissions changes on respiratory hospital admissions with the analysis restricted to days with higher temperatures. To define a ‘hot day’, studies typically consider different percentiles of the annual distribution of daily mean temperatures, and we selected here the 95^{th} percentile to define a ‘hot day’ (as also used by Pattenden et al. (2010)). Taking the Central England Temperature (CET) as an indicator for UK temperature (Parker et al., 1992), gives 18.4°C as the 95^{th} percentile of the annual distribution of daily mean temperature (with a total of 34 days above this temperature in 2018). During summer when O_3 concentrations are generally higher, the impact of emission reductions is to lower some of the peak O_3 values (Fig. 2b), and results in a reduction in respiratory hospital admissions associated with O_3 exposure for these days (Table 3). This would indicate that while the overall annual total of respiratory emergency hospital admissions across the year increases in response to anthropogenic emissions changes (Fig. 4, Table 1), those admissions occurring on the hottest days (when O_3 concentrations are highest) are actually reduced, demonstrating a benefit of anthropogenic emission controls on hot days.

Table 3: Health impact assessment restricted to hot days (>95th percentile of annual daily mean CET distribution) at the UK level. (The total number of days in consideration here is 34, to consider when comparing to annual totals. Population changes are not assumed here).

Emission year	2018	2030	2040	2050
O_3 exposure (μg m^{-3})	86.1	85.1	84.2	84.0
Mean attributable fraction	6.2%	6.2%	6.1%	6.1%
Total attributable hospital admissions over the 34-days (change from 2018)	4958	4904 (−54)	4851 (−107)	4841 (−116)
Mean daily attributable hospital admissions (change from 2018)	146	144 (−1.6)	143 (−3.1)	142 (−3.4)

4 Discussion

Our results suggest that current policies driving anthropogenic emission changes will lead to an increase in future UK O_3 concentrations and hence in associated respiratory emergency hospital admissions; for example, by 4.6% in 2050, an increase of 2,801 admissions across the UK from 2018 assuming no population changes. Hospital admissions increase by 11.7% by 2050 when population projections are also included (Table S3). Previous work has shown that in future there are likely to be improvements in UK air quality through reductions in $PM_{2.5}$ and NO_2, and a reduction in the mortality burden associated with long-term exposure (Macintyre et al., 2023), whilst here we show that there may be a concurrent, albeit smaller, increase in respiratory hospital admissions. When restricting the analysis to the hottest days of the year, future anthropogenic emission reductions result in lower O_3 concentrations on hot days, and lower associated respiratory hospital admissions. While the overall annual burden of associated respiratory hospital admissions increases (as hospital admissions are generally greater in colder months, and the effect of emission reductions is generally to increase O_3 concentrations in these months), on hot days O_3 concentrations are lower in simulations with lower anthropogenic emissions, which could be beneficial for health and care services during periods of hot weather (Table 3). The use of different meteorological years in our analysis results in only very small (<1%) differences in the annual total attributable daily hospital admissions, although there is a clear influence of meteorology on day-to-day variation in O_3 concentrations and thus in associated hospital admission (Fig. 5).

It is not possible to know future meteorology exactly, but clearly there will be some potential impact of interannual variability in meteorology on future O_3 concentrations and health burdens as well. As heatwaves (commonly associated with O_3 episodes) are projected to become more frequent and intense with climate change in the UK (Murphy et al., 2018), there may be implications for response planning when periods of hot weather are forecast, as there may also be health effects from increased O_3. While the aim here is to examine the impact of changes in anthropogenic emissions, the role of natural emissions of O_3 precursors (for example BVOC emissions that increase with temperature), and hemispheric CH_4 may become more important in future as anthropogenic emissions become smaller, and as temperatures rise. BVOC emissions are included in the model and are dependent on the meteorology; though as the aim of the study is to evaluate the impacts of anthropogenic air pollutant emission changes associated with current policies, we use the same meteorological year to aid comparison. However, whilst BVOC emissions are the same for simulations using the same meteorological year, the impact of the BVOC emissions on O_3 formation will vary with the different anthropogenic VOC and NO_x emissions. Our simulations extend to 2050 which has been shown to be about the point where future temperature projections start to significantly diverge for different scenarios. It is not possible to simulate future meteorology at the high spatial and temporal resolution, and over a large enough spatial domain, required to drive BVOC emissions and other meteorologically-dependent transboundary atmospheric chemistry processes within air quality models such as the EMEP4UK model we use here. Instead, our selection of a relatively hot recent year (2018) that may become a 50-50 likely summer by 2050 (as noted in the UKCP18 headline findings report) will emulate potential representation of future conditions.

Regional variation in the impacts of emissions changes is generally modest; the largest impacts of emissions changes are observed in London, which has the greatest reduction in NO_x emissions in future (leading to enhanced O_3 through reduced titration with NO). At the daily scale, the largest changes are to the lowest O_3 concentrations which are due to high NO chemically depleting the O_3. By reducing NO emissions, the most polluted NO days are reduced in future, leading to increases in O_3 on these days (Fig. 3). This effect has also been observed during the sudden curtailment of emissions, particularly from transport, during the 'lockdown' periods associated with COVID-19 measures in 2020 (AQEG, 2020; Grange et al., 2021; Jephcote et al., 2021).

Our results here align with other studies assessing the impact of future emission changes on O_3 concentrations which find similar increases near population centres, driven by NO_x emission reductions (Heal et al., 2013; Hedegaard et al., 2013; Fenech et al., 2021). Very few studies have quantified hospital admissions associated with O_3 across the UK (with the majority examining mortality); our results for 2018 (60,488) are larger than those of Heal et al. (2013, who reported 30,700 respiratory hospital admissions associated with O_3 exposure in 2003, though our study has higher baseline admission rates, different background emission scenarios and atmospheric model resolution, and a slightly larger ERF coefficient; our study also has higher O_3 levels, with averages in the range 70-80 μg m^{-3} compared with 60-70 μg m^{-3}. Stedman et al. (1997) estimate about 6% of respiratory emergency hospital admissions are attributable to O_3, which compares with the figures in our study, noting this is strongly affected by use or not of an O_3 concentration threshold, where health effects are assumed to be attributed to the exposure only above a given threshold for effects, and no health impacts attributed at concentrations below this. The choice of exposure coefficient and concentration threshold for effects is important as this impacts the overall health burden estimates, which are higher when no threshold is used (e.g. Stedman 1997 found a 20-fold increase in burden estimate when using no threshold as opposed to a 50 ppbv threshold). However, when assessing the impact of O_3 changes, the impact of the change appears larger when a threshold is used; for example if a threshold for O_3 effects is assumed, health burdens are more sensitive to changes in O_3 concentrations, although total health burdens are roughly an order of magnitude lower (Stedman et al., 1997; Stedman and Kent, 2008; Heal et al., 2013; COMEAP, 2022). Recent recommendations for the UK are not to use a threshold for effects (COMEAP, 2022), which we followed in this work. There is some evidence that temperature may act as a modifier for O_3 effects, and this may be important in future if heatwaves and episodes of O_3 become more common or intense (Pattenden et al., 2010; Atkinson et al., 2012). It is not possible to predict future changes in exposure-response relationships. Changes in baseline hospital admissions may also influence overall health impact estimates.

The UK has an objective value of 100 μg m^{-3} for the daily maximum 8-hour mean O_3, not to be exceeded more than 10 times a year as set out in the National Air Quality Objectives as part of the Air Quality Standards Regulations 2010 (UK Government, 2010). Our study shows that, in future, the number of days exceeding the target value of 100 μg m^{-3} is projected to increase in London (from 44 in 2018 to 57 days in 2050), North West (26 in 2018 to 36 in 2050), and East England (37 in 2018 to 48 in 2050), and decreases in others (1 or 2 fewer days in Scotland, Northern Ireland and the South West), with small increases in other areas. In 2021, the WHO updated its recommended air quality guideline (AQG) levels for air pollutants, including O_3, with a new guideline of 100 μg m^{-3} for 8-hour mean O_3 (reduced from 120 μg m^{-3}) not to be exceeded 3-4 times per year (World Health Organization, 2021). Our analysis shows it may be harder for the UK to meet its objective and the new WHO AQG values in future. We note however that the data presented above are based on population-weighted mean values for the region or country and that changes in daily maximum 8-hour mean O_3 at specific locations will vary.

At global scale, surface O_3 is expected to decrease slightly in future in low-NO_x regimes, due to increased humidity leading to greater O_3 destruction, whereas in polluted regions, model studies suggest an increase in O_3 (Szopa et al., 2021), with the influence of climate changes on air quality being dependent on the particular emissions scenarios (Fiore et al., 2015). Here we use fixed CH_4 and O_3 boundary conditions, and while wider hemispheric trends in ozone from global trends in emissions of CH_4 will impact on UK concentrations, the direction of change is unclear (AQEG, 2021).

A previous study using the same modelling as here showed that reducing emissions generally improves air pollution ($PM_{2.5}$ and NO_2) and the associated mortality burdens by around a third by 2050 (Macintyre et al., 2023), though as no threshold for effects is assumed, health impacts still remain. Here however we show that there can be some worsening effects related to secondary pollutants such as O_3, though for hot days (that often coincide with higher O_3) these may be reduced. While the overall impacts of emission reductions are beneficial for health (Macintyre et al., 2023) (acknowledging that it is challenging to directly compare mortality burdens and daily hospital admissions), we find that there may be increased effects on emergency respiratory hospital admissions, which is important for health system response planning when such episodes are forecast. Our analysis in this study and a previous study (Macintyre et al., 2023) provides quantitative estimates of the potential effects to health of current policy commitments. The methods here use a simple burden calculation to estimate health effects, though the specific impacts will depend on the speed and scale of emission reductions. Our results indicate the likely impacts to health if emissions likely to be reached at a particular indicative year in the future were to be met and sustained. While we examine the impact of population size changes, potential changes in baseline hospital admission rates are another source of uncertainty for future projections. An important point is that the UK population is ageing; as the likelihood of living with chronic disease and susceptibility to illness in general tends to increases with age, there may be increases in the baseline rates of many health effects associated with air pollution. Future baseline rates of hospital admissions are challenging to predict in any robust manner as many factors influence these (e.g. circulating respiratory infections, rates of chronic respiratory disease, influenza vaccine uptake and effectiveness) and thus are not considered here, which is a limitation of the study. Another limitation is related to potential future changes in exposure-response relationships; No known methodology exists to reliably predict these as these depend on multiple factors such as the future air pollution composition, exposure estimates, and population health, thus we do not examine this in the current analysis. Finally, the effects of higher air pollution levels and air pollution episodes will undoubtedly be modified by behaviours that affect exposure and inhaled dose, such as time spent indoors and exercise undertaken.

Conclusion

The effect of future emission policy on air pollution in the UK has been modelled and a quantitative HIA to estimate impacts on emergency respiratory hospital admissions performed. While our previous work showed significant reductions in the mortality burden due to lower $PM_{2.5}$ and NO_2 concentrations, we estimate here that there will be a small increase in O_3 exposure in future, and hence in associated hospital admissions, as a result of lower NO in urban areas, where most people reside. The main impacts on O_3 occur on the days with lowest O_3 (i.e. highest NO_x days) in regions that are already more polluted generally, such as London. Emission control policy has clear benefits to health through reducing $PM_{2.5}$ and NO_2, though effects related to secondary pollutants such as O_3 can be complex and should be acknowledged. As other air pollutants in the UK continue to decline thanks to emission controls, it seems likely that the importance of O_3 will grow. Future work on the health impacts of air pollution episodes in summer coinciding with heat should consider potential worsening effects associated with local increasing O_3 concentrations, and may need to be considered when planning for public health protection during these periods.

Acknowledgements

This research was partly funded by the National Institute for Health Research (NIHR) Health Protection Research Unit (HPRU) in Environmental Change (NIHR200909), a partnership between the UK Health Security Agency and the London School of Hygiene and Tropical Medicine, University College London, and the Met Office, and the HPRU in Environmental Exposures and health, a partnership between the UK Health Security Agency and the University of Leicester. The views expressed are those of the authors and not necessarily those of the NIHR, the UK Health Security Agency, or the Department of Health and Social Care. We acknowledge health data from ONS, Public Health Scotland, Northern Ireland Statistical and Research Agency (NISRA), dhcw.nhs.wales, and population data from the National Population Database, and UK Centre for Ecology & Hydrology. The work of the UK Centre for Ecology & Hydrology was supported through funding from the Department for Environment, Food and Rural Affairs under the contract “Research & Development Support for National Air Pollution Control Strategies (ECM: 62041) 2021 to 2024” and builds upon research supported by the Natural Environment Research Council award number NE/R016429/1 as part of the UK-SCAPE programme delivering National Capability. CH is supported by a NERC fellowship (NE/R01440X/1) and acknowledges funding for the HEROIC project (216035/Z/19/Z) from the Wellcome Trust.

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