

1 **Future impacts of O₃ on respiratory hospital admission in the UK health from current**
2 **emissions policies**

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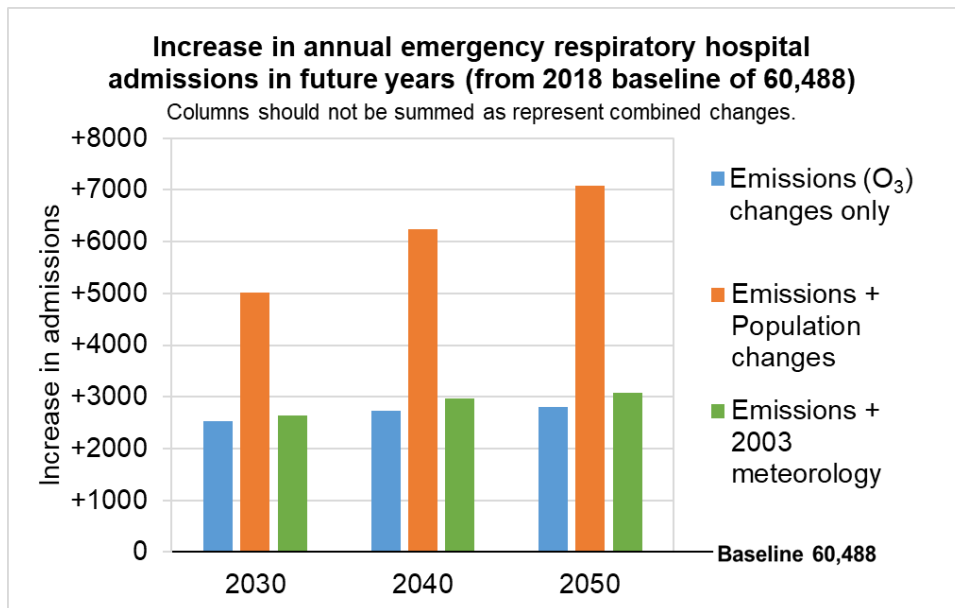
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20 **Highlights**

- 21 • Policies on anthropogenic emissions dominate health impacts of air pollution
- 22 • O₃ simulated at 3 km resolution for 2030, 2040 and 2050 emissions pathways
- 23 • Hospitalisations calculated with latest UK health impact assessment
24 recommendations
- 25 • 7078~~Small~~ increase in respiratory hospitalisations by 2050 for current policy c.f. 2018
- 26 • Estimated increases are 2-3 times lower if population growth is not included

27
28 **Graphical abstract**



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31

32 **Keywords**

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

34

35 **Abstract**

36 Exposure to ambient ozone (O₃) is associated with impacts on human health. O₃ is a
 37 secondary pollutant whose concentrations are determined inter alia by emissions of
 38 precursors such as oxides of nitrogen (NO_x) and volatile organic compounds (VOCs), and
 39 thus future health burdens depend on policies relating to climate and air quality. While
 40 emission controls are expected to reduce levels of PM_{2.5} and NO₂ and their associated
 41 mortality burdens, for secondary pollutants like O₃ the picture is less clear. Detailed
 42 assessments are necessary to provide quantitative estimates of future impacts to support
 43 decision-makers. We simulate future O₃ across the UK using a high spatial resolution
 44 atmospheric chemistry model with current UK and European policy projections for 2030,
 45 2040 and 2050, and use UK regional population-weighting and latest recommendations on
 46 health impact assessment to quantify respiratory emergency hospital admissions associated
 47 with short-term effects of O₃. We estimate 60,488 admissions in 2018, increasing by 4.2%,
 48 4.5% and 4.6% by 2030, 2040 and 2050 respectively (assuming a fixed population).
 49 Including future population growth, estimated emergency respiratory hospital admissions are
 50 8.3%, 10.3% and 11.7% higher by 2030, 2040 and 2050 respectively. Increasing O₃
 51 concentrations in future are driven by reduced nitric oxide (NO) in urban areas due to
 52 reduced emissions, with increases in O₃ mainly occurring in areas with lowest O₃
 53 concentrations currently. Meteorology influences episodes of O₃ on a day-to-day basis,

54 although a sensitivity study indicates that annual totals of hospital admissions are only
55 slightly impacted by meteorological year. While reducing emissions results in overall benefits
56 to population health (through reduced mortality due to long-term exposure to PM_{2.5} and
57 NO₂), due to the complex chemistry, as NO emissions reduce there are associated local
58 increases in O₃ close to population centres that may increase harms to health~~should be~~
59 ~~considered in future health advice and planning.~~

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61

62 **1 Introduction & background**

63

64 Air pollution has detrimental impacts on human health. The main air pollutants associated
65 with impacts on human health are PM_{2.5} (particulate matter with aerodynamic diameter
66 <2.5 µm), nitrogen dioxide (NO₂), and ozone (O₃), with long-term exposure to ambient PM_{2.5}
67 and NO₂ in the UK estimated to have an effect equivalent to 29,000–43,000 deaths annually
68 (Mitsakou et al., 2022). Exposure to ambient O₃ is also associated with ill health, particularly
69 respiratory effects including exacerbation of asthma symptoms.; O₃ is estimated to
70 contribute to over 5 million disability adjusted life years (DALYs) lost annually globally
71 (Forouzanfar et al., 2015). The evidence of O₃ impacts on health is strongest for
72 associations between short-term exposure (as quantified using the daily maximum 8-hour
73 running mean O₃) and respiratory and cardiovascular hospital admissions, though with more
74 uncertainty for the latter (COMEAP, 2015). Quantification of the impact on health of long-
75 term exposure to O₃ is not recommended by the UK's Committee on the Medical Effects of
76 air Pollutants (COMEAP) as there is currently insufficient evidence (COMEAP, 2015;
77 Huangfu and Atkinson, 2020; COMEAP, 2022).

78

79 Ozone is not directly emitted but is a secondary pollutant formed through complex
80 (photo)chemical reactions in the atmosphere involving NO_x (= NO + NO₂) and volatile
81 organic compound (VOC) emissions, including of methane (CH₄). The levels of methane
82 contribute to the longer-term (annual average) concentrations of O₃ that are also part of O₃
83 exposure and upon which the shorter-term variations in O₃ driven by non-methane VOCs are
84 superimposed. Concentrations of O₃ are also affected by meteorology, which influences its
85 build-up through photochemistry and air mass transport and dilution, and also via the rate of
86 O₃ deposition to the surface (Royal Society, 2008). High levels of NO_x (e.g. close to emission
87 sources) also act to limit O₃ formation, which is why reductions in NO_x emissions in urban
88 areas, largely associated with improvements in vehicle exhaust emission standards, have
89 led to modelled local increases in O₃ (Carnell et al., 2019).

90

91 Air pollution levels have generally improved in the UK over the past several decades, thanks
92 to environmental and public health initiatives. Carnell et al. (2019) reported that reductions in
93 air pollution levels between 1970 and 2010 led to a reduction in UK attributable mortality
94 associated with exposure to PM_{2.5} and NO₂ of 56% and 44% respectively; however, the
95 same study reported an increase in O₃-attributable respiratory mortality of 17%. This reflects
96 a different trend in response to recent air pollutant emissions changes for O₃ in urban areas,
97 where most people live, than for urban NO₂ and PM_{2.5} (AQEG, 2021). In particular, as noted
98 above, there is potential for local increases in O₃ due to reductions in emissions of NO_x, with
99 a recent report noting upwards trend in urban ozone of 5 – 9 µg m⁻³ between 2000-2019
100 (AQEG, 2021). A modelling study on possible UK net-zero policies showed that while longer-
101 term (April to September) mean O₃ concentrations may reduce, short-term O₃ metrics
102 revealed less O₃ suppression from reduced NO_x, particularly in winter (Williams et al., 2018).
103 The effect of reduced traffic NO_x emissions increasing urban O₃ was also illustrated during
104 the travel restrictions implemented as part of COVID-19 'lockdown' restrictions (AQEG,
105 2020; [Lee et al., 2020](#); [Jephcote et al., 2021](#)).

106

107 Changes in climate will also modify concentrations of surface O₃, as it will for other air
108 pollutants (von Schneidmesser et al., 2015; Doherty et al., 2017). However, the Doherty
109 et al. review concludes that for O₃ over Europe in the next few decades, changes in
110 anthropogenic emissions of the VOC and NO_x precursors will have much greater influence
111 on O₃ levels than changes in climate. For example, model simulations of impacts of climate
112 scenarios and compatible air pollutant emissions under CMIP5 scenarios showed O₃
113 exposure metrics over Europe in 2050 change by 3% (±8%) to 5% (±11%) due to changes in
114 climate, but by -24% (±10%) to -43% (±7%) due to emission changes (Colette et al., 2013).
115 Over the UK, changes in annual mean O₃ by 2030 (compared to 2003) due to changes in
116 precursor emissions (-3.0 to +3.5 ppbv) are larger than for a +5°C scenario (+1.0 to
117 +1.5 ppbv) [including temperature effects on biogenic VOCs, chemical reaction rates and O₃
118 dry deposition](#), with the latter being comparable to the changes in O₃ arising from normal
119 interannual meteorological variability (Heal et al., 2013). Resulting changes in mortality
120 burdens were 4.1% associated with O₃ changes due to +5°C, whereas they were 15.6% to
121 27.7% for emissions changes (depending on emission scenario), assuming no threshold for
122 effects (Heal et al., 2013).

123

124 In a previous study we performed an up-to-date quantitative impact assessment of the
125 mortality burdens associated with long-term effects of PM_{2.5} and NO₂ in 2030, 2040 and
126 2050 using a high spatial resolution atmospheric chemistry and aerosol model with the latest
127 official UK and European emissions datasets (Macintyre et al., 2023). We found that

128 mortality burdens associated with these two pollutants were reduced by over 30% beyond
129 2030. As O₃ is also impacted by such emission policies, but in more complex ways as
130 described above, we expand our analysis here to consider impacts on respiratory hospital
131 admissions associated with short-term exposure to O₃, using a similar health impact
132 assessment (HIA), to more fully capture future impacts to health associated with changing
133 emissions.

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135

136 **2 Methods**

137

138 ***2.1 Air pollution modelling and emissions***

139 Simulations of UK present-day and future air quality at 3 km × 3 km spatial resolution and
140 hourly temporal resolution were undertaken using the EMEP4UK¹ atmospheric chemistry
141 transport model version rv4.36. The chemical model was driven by hourly meteorology from
142 the Weather Research and Forecasting (WRF) model version 4.2.2 (wrf-model.org). The
143 EMEP4UK-WRF modelling system has been widely used to simulate historic, present-day
144 and potential future air quality over the UK (Heal et al., 2013; Vieno et al., 2014; Vieno et al.,
145 2016; Nemitz et al., 2020), including provision of evidence on air quality to the UK
146 government (HPA, 2012; AQEG, 2013; AQEG, 2017; AQEG, 2021). Details of the model
147 system have been described previously in our companion HIA study (Macintyre et al., 2023)
148 and in the above references, and are also detailed in the Supplementary Material, so only a
149 brief description of the model runs used in this study is provided here.

150

151 Anthropogenic emissions of NO_x, NH₃, SO₂, CO, NMVOC (non-methane VOC), PM_{2.5} and
152 PM_{co} (coarse particulate matter, aerodynamic diameter >2.5 µm and <10 µm) for the 2018
153 simulation for the UK were taken from the National Atmospheric Emissions Inventory
154 (naei.beis.gov.uk). UK emissions for model simulations of atmospheric composition in 2030,
155 2040 and 2050 use the 'business as usual (BAU)', also referred to as 'baseline' scenarios
156 developed for Defra (ApSimon et al., 2019; Defra, 2022; Oxley et al., 2023) which reflect
157 assumed trends for anthropogenic emissions under existing interventions and policies
158 relating to air quality, including adjustments to include the projected impact of recent policy
159 not included in the 2018 NAEI projections. Emissions projections for the rest of the European
160 domain use official anthropogenic EMEP emissions fields for the corresponding years
161 (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).

162 evaluate the impact of changes in anthropogenic emissions on changes in health burden.
163 Biogenic VOC (BVOC) emissions are included in the model and are dependent on the
164 meteorological year (here held constant to evaluate changes in anthropogenic emissions),
165 and thus will be the same in simulations using the same meteorological year. Additionally, a
166 sensitivity experiment was run for future years using 2003 meteorology, as an example of a
167 hot year in the UK for comparison. The CH₄ concentration and the O₃ concentrations at the
168 boundary of the extended European domain were fixed at 2018 values.

169

170 **2.2 Exposure metric and population-weighting**

171 The daily maximum of the 8-hour running mean O₃ concentration was calculated from
172 gridded hourly model output for each of the simulation years at the native grid resolution, to
173 match the O₃ exposure metric used in the epidemiology (see equations in Section 2.3). This
174 O₃ exposure metric was population-weighted to create regional estimates for the nine
175 regions in England, and for Scotland, Wales and Northern Ireland (Fig. 1b) using 100 m
176 gridded residential population information for England, Scotland and Wales (National
177 Population Database, 2020), and at 1 km for Northern Ireland (Reis et al., 2017).

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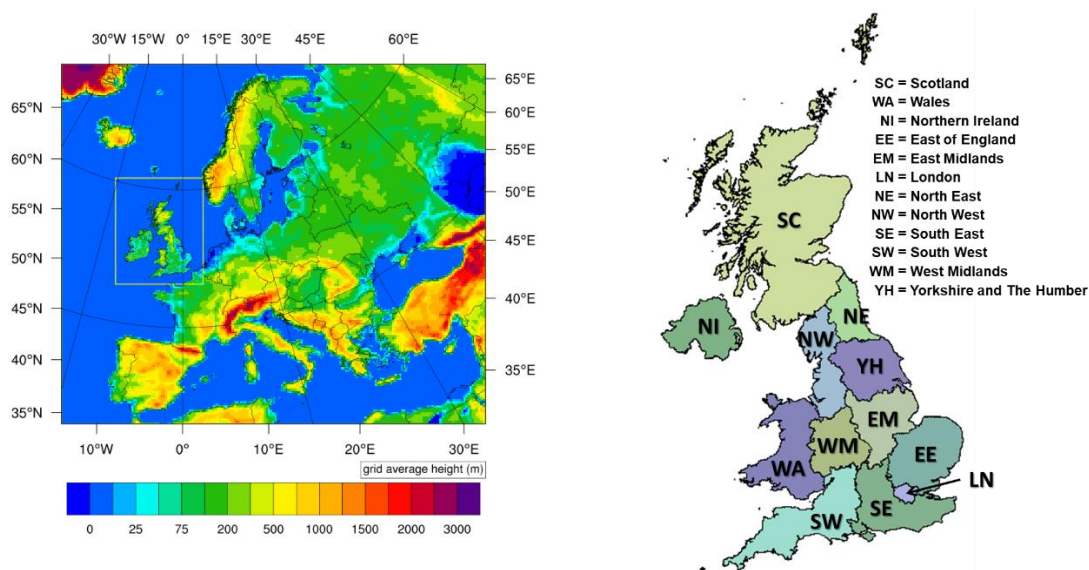


Figure 1. (a) The EMEP4UK model domain showing the 3 km x 3 km resolution British Isles domain (yellow box) nested within an extended Europe domain at 27 km x 27 km (plotted variable is altitude). (b) Regions used in the health impact assessment analysis.

179

180 **2.3 Health impact calculations**

181 The hospital admissions associated with short-term exposure to O₃ (H_{aq}) were estimated as
182 follows: $H_{aq} = H_T \times AF$, where H_T is the total emergency respiratory hospital admissions (all-
183 ages) in the region, and AF is attributable fraction of the health outcome in each region
184 associated with exposure to O₃. AF is calculated based on the percent increase in

185 emergency respiratory hospital admissions per 10 $\mu\text{g m}^{-3}$ O_3 reported from meta-analyses of
186 time-series studies. We used an exposure-response coefficient corresponding to an increase
187 in emergency respiratory hospital admissions (all-ages) of 0.75% (95% CI: 0.30%, 1.2%) per
188 10 $\mu\text{g m}^{-3}$ daily maximum 8-hour running mean O_3 with no threshold for effects (COMEAP,
189 2015; COMEAP, 2022).

190

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

199

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

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212 **3 Results**

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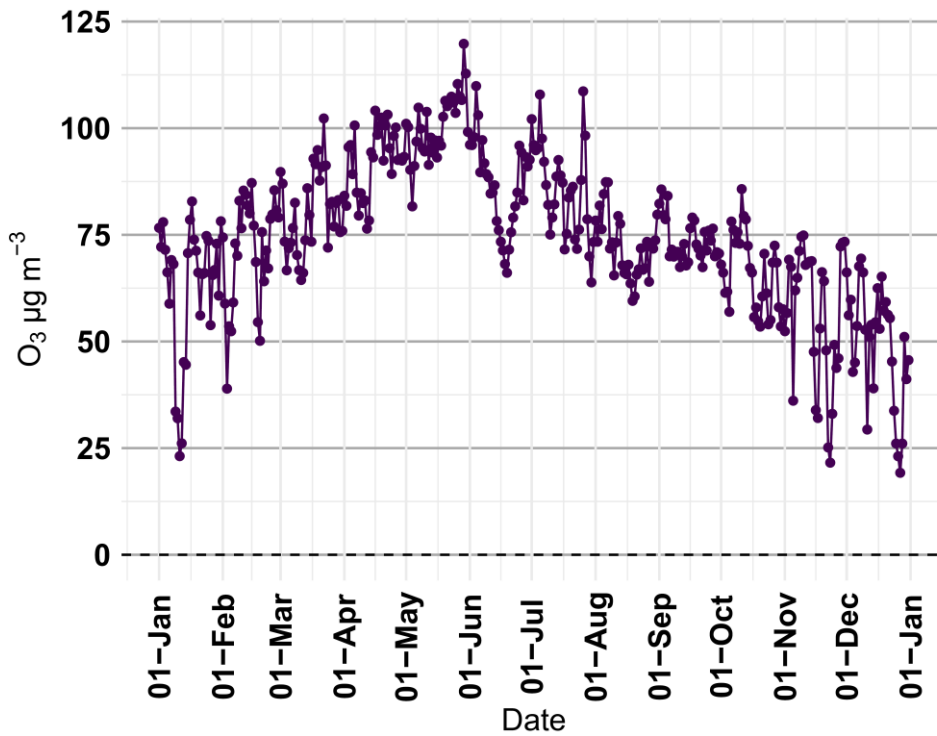
214 ***3.1 Impact of future emissions on O_3 and associated hospital admissions***

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

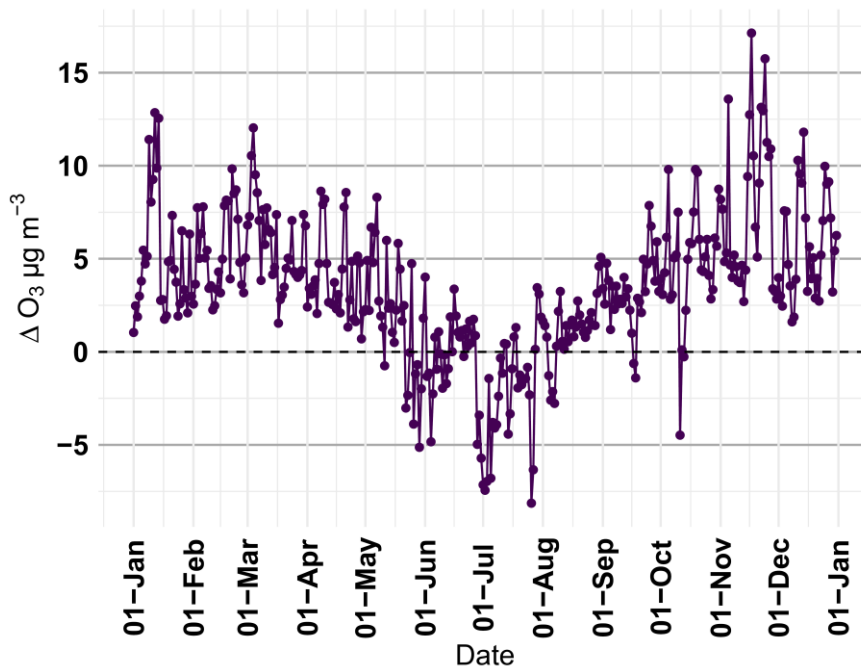
² The use of the term "interim" in the 2020 release reflects the interval between the 2020-based principal projection and subsequent projections, which will incorporate Census 2021 data. It also recognises uncertainties in the mid-2020 base year and in setting long-term demographic assumptions following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

218 of UK O₃ (AQEG, 2021). The lowest values appear mostly in the London region, with the
 219 West Midlands, East of England, and North west also showing low excursions (Fig. 3).
 220 Comparing the impact of 2050 emissions against 2018 emissions shows an overall increase
 221 in O₃ (Fig. 2b,c). The increases in O₃ under 2050 emissions are in areas where O₃ is lowest
 222 with 2018 emissions, thus thewith greatest increases in O₃ 2050 are in those regions
 223 showing lower values in 2018 (Fig. 3). In winter months (November to March) the O₃
 224 concentrations increase in response to emission changes across all regions, whereas for
 225 warmer months the changes are mixed, with some increases and some decreases (Fig 2b).
 226
 227
 228

(a) 2018 O₃ exposure (population-weighted value for the UK as a whole)



(b) Absolute difference in 2050 O₃ exposure with respect to 2018 O₃ exposure
 (i.e., positive indicates higher O₃ within 2050 emissions as compared with 2018)



(c) Relative difference in 2050 O₃ exposure with respect to 2018 O₃ exposure

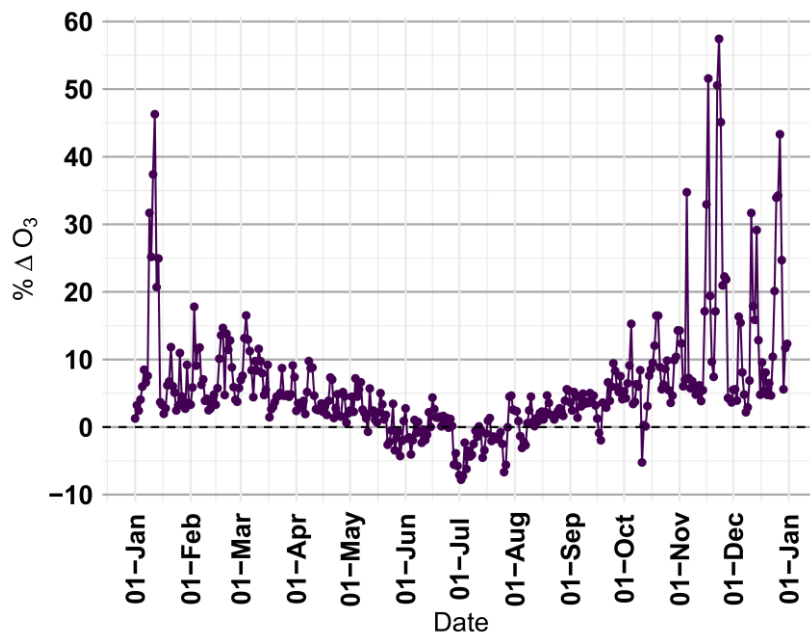
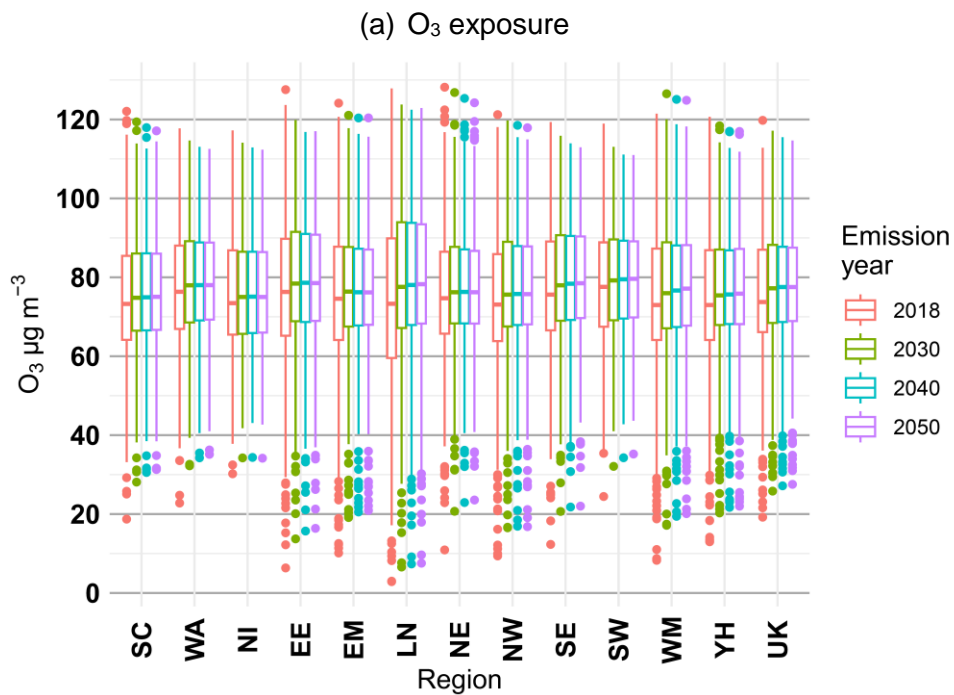


Figure 2: (a) ~~Regional-UK~~ population-weighted O₃ exposure (daily maximum 8-hour running mean) ~~by country and regions in England~~ in 2018; (b) Difference in 2050 compared to 2018; (c) percent difference in 2050 compared to 2018. ~~(See Fig. 1b for region codes.)~~

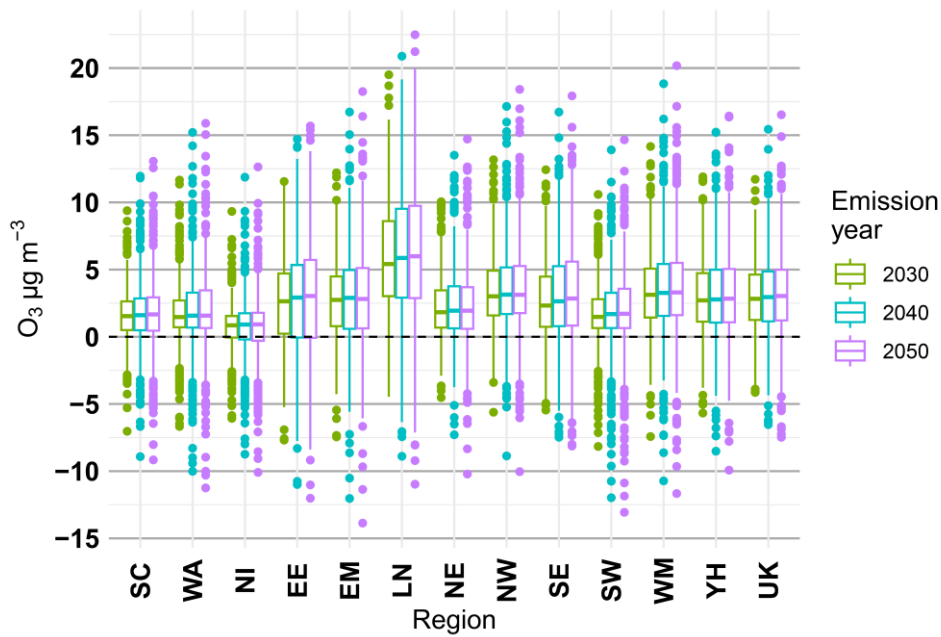
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230 Examining the overall annual distribution of O₃ exposure ~~for each of the~~ regions and
 231 emission years (Fig. 3) reveals small increases in median daily O₃ exposure, of around 2 –
 232 4 μg m⁻³, and up to 6 μg m⁻³ in London, with some individual days showing >20 μg m⁻³
 233 increase (Fig. 3b). ~~There is with~~ a narrowing of the range of exposures by 2050 (Fig. 3a), as
 234 high values in summer are slightly reduced, and low values in winter months are increased

235 (Fig. 2b). While the range of exposure narrows in future, the range in the difference
236 compared to 2018 appears to expand (Fig. 3b) indicating less variations in the daily
237 concentrations; this is due to the decreases in O₃ occurring where there is existing higher O₃
238 (so peak O₃ values in summer are reduced), and the increases in O₃ being where low O₃
239 values currently occur, particularly so for the instances of very low O₃ values. As the
240 increases in O₃ (due to less suppression by NO) occur on background values that are
241 already small, this leads to very high apparent changes when expressed as a percentage
242 (Fig. 3c).
243



(b) absolute differences in O₃ exposure compared to 2018



(c) relative differences in O₃ exposure compared to 2018

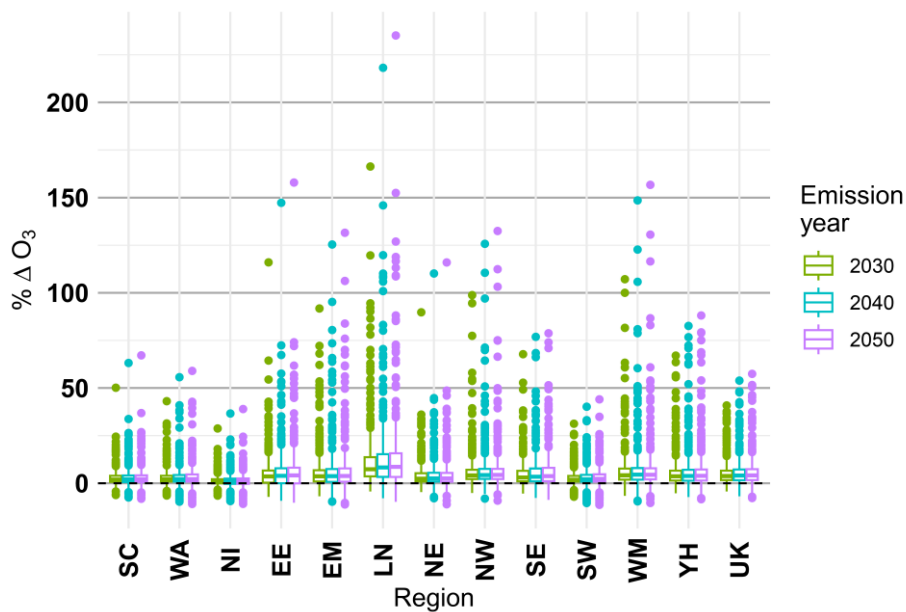


Figure 3: Boxplots* of daily population-weighted O₃ exposure (based on daily max 8-hour running mean) across all regions and different emission years (meteorology is for 2018 in all simulations). (a) Daily absolute values; (b) daily absolute differences compared to 2018; (c) percent differences compared 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.

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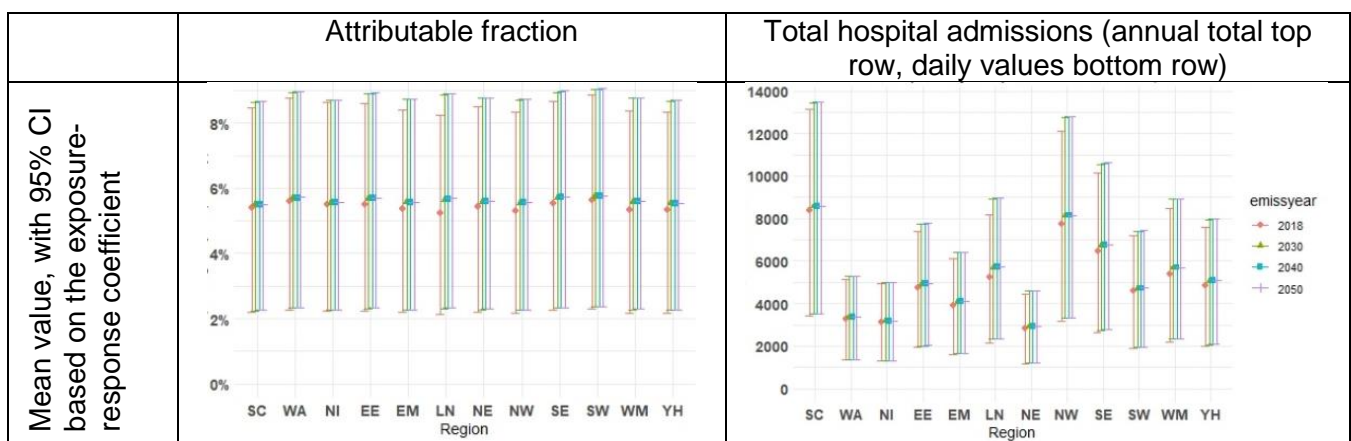
The annual mean daily ~~attributable~~ fraction of emergency respiratory hospital admissions attributable to short-term effects of O₃ exposure was 5.4% in 2018 (with mean ranging from

247 5.25% in London to 5.65% in the South West), rising slightly to 5.6% by 2050 (range 5.52%
 248 in Scotland to 5.78% in the South West) (Fig. 4). The annual cycle of the attributable
 249 fractions is in Supplementary Fig. S1. While attributable fractions are driven by air pollution
 250 concentrations, the absolute totals of attributable hospital admissions are also influenced by
 251 baseline admission rates and population in each region. Northern Ireland, Scotland, and the
 252 North West have higher baseline admission rates, and the South East, London and the
 253 North West have higher populations, leading to the highest overall attributable hospital
 254 admissions being estimated in Scotland, North West and South East (Fig. 4). Given that the
 255 changes in future O₃ concentration are comparatively small, the differences in annual
 256 attributable fractions and hospital admissions in future years compared with 2018 shown in
 257 Fig. 4 are likewise relatively modest. (The values of attributable fractions and hospital
 258 admissions using the confidence interval range on the exposure response function are
 259 shown in Supplementary Fig. S2.)

260

261 Our analysis suggests that for 2018 the annual total of daily emergency hospital admissions
 262 for respiratory causes across the UK associated with short-term exposure to O₃ was 60,488
 263 (95% CI: 24,673 – 94,927), and that this will be higher by 4.2%, 4.5%, and 4.6% in 2030,
 264 2040 and 2050 respectively (relative to 2018) reaching 63,289 (95% CI: 25,827 – 99,278)
 265 with 2050 emissions, an increase of 2,801 (1,154 – 4,352) (Table 1). The overall estimated
 266 associated hospital admissions in each region are also influenced by baseline admission
 267 rates and population size in each region, with the North West, London and South East
 268 having larger populations, and Scotland having larger admission rates.

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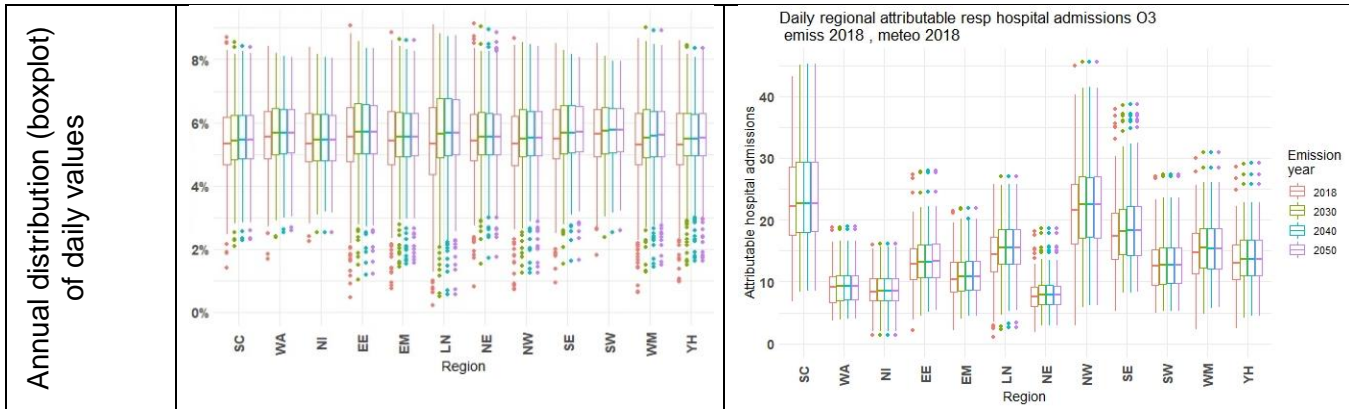


Figure 4: Regional attributable fraction (left column) and annual total respiratory emergency hospital admissions (right column) associated with short-term exposure to O₃. The top row shows annual totals, with error bars showing the 90% confidence interval on the exposure-response coefficient. The bottom 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.)

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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 (25,8270 – 99,278)

272

273 When accounting for changes in population size in the future (scaled based on the
 274 population changes in Table S2), the annual total of daily hospital admissions for the UK in
 275 2050 rises from 63,289 to 67,566 (+4,277) (Table 2), meaning that the increases from the
 276 2018 baseline of 4.2%, 4.5%, and 4.6% in 2030, 2040 and 2050 respectively (due to O₃
 277 changes only), are now increased to 8.3%, 10.3% and 11.7% respectively (due to both O₃
 278 and population changes) (full breakdown of results in Table S3). This total hides some
 279 regional variation, as although overall UK population increases by 2050, the population in
 280 Scotland is expected to rise in 2030, but then fall thereafter and by 2050 is 1.9% lower than
 281 its 2018 value (Table S2). The increase in hospital admissions in Scotland (due to O₃ only
 282 with no population change) of 222 by 2050, becomes an increase of 55 hospital admissions
 283 over the same period when accounting for the shrinking population. In Northern Ireland,
 284 population is projected to increase in 2030 and 2040, but to reduce in 2050 closer to 2030
 285 levels (Table S2). In all other regions, population changes increase the health burden, with
 286 the greatest combined (O₃ and population) changes in hospital admissions by 2050 being in
 287 London (+19.1%), the West Midlands (+14.9%), and North West (+14.7%) with respect to
 288 the 2018 baseline.
 289

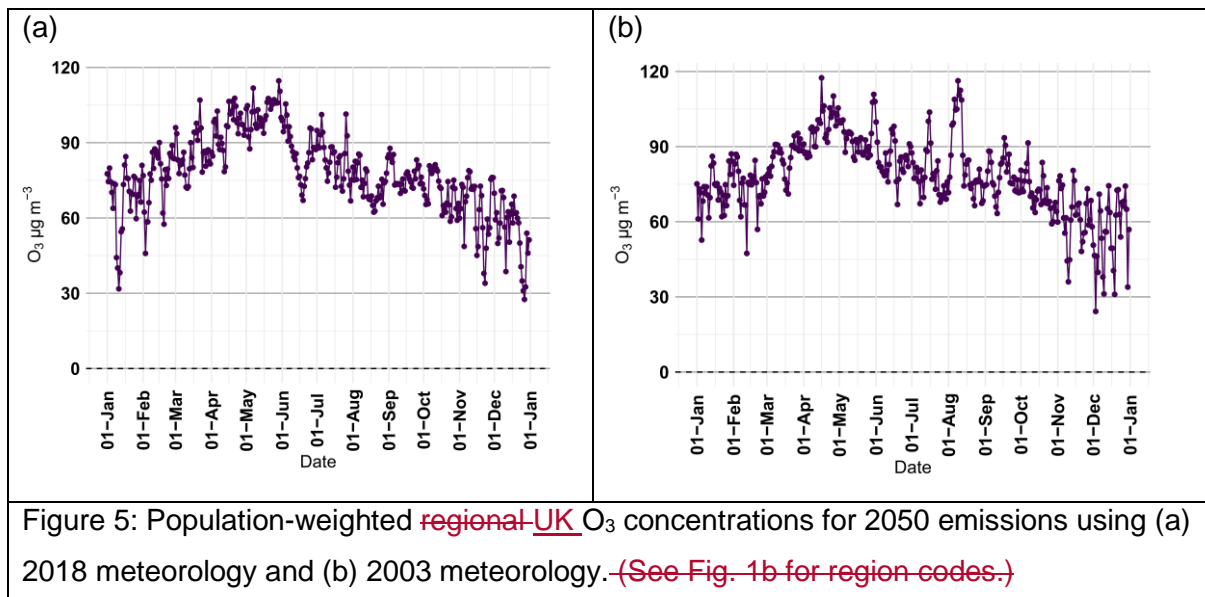
Table 2: Respiratory emergency hospital admissions associated with O ₃ for future years, with and without projected population size changes.							
Region	2018	2030		2040		2050	
		Base	With population change	Base	With population change	Base	With population change
England	45,698	47,907	50,101	48,094	51,348	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

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 291

292 **3.2 Impact of inter-annual variation in meteorology on O₃ and hospital admissions**

293 The future emission years were also run with 2003 meteorology as another historical ‘hot’
 294 year for the UK as a sensitivity study for the impact of interannual meteorology on O₃
 295 concentrations. While the annual means of future population-weighted daily maximum 8-
 296 hour mean concentrations are very similar when using either 2018 or 2003 meteorology
 297 (less than 1 µg m⁻³ difference on a UK mean of all such values of 78 µg m⁻³), there are
 298 differences in the annual cycles of the daily concentrations, as illustrated in Fig. 5 for 2050
 299 emissions. Overall, the annual cycles both show higher concentrations in April and May (as

300 expected), with some peaks during heatwaves and anticyclonic airflows; for example the
 301 effects on simulated O₃ of using the meteorology associated with the European heatwave in
 302 early August 2003³ are clearly visible (Fig. 5b). There is also a particular O₃ peak associated
 303 with the 16 April 2003 meteorology, coinciding with an unusually hot day for that time of year
 304 (26.7°C was recorded in Northolt, Greater London) with clear skies and south easterly air
 305 flow, with population-weighted daily maximum 8-hour mean O₃ exceeding 120 µg m⁻³ in
 306 West Midlands, North West, East Midlands, and Yorkshire and The Humber ([Supplementary](#)
 307 [Fig. S4](#)). May 2018 was the sunniest May on record, leading to high O₃ values when using
 308 that month's meteorology, followed by a decline in O₃ in early June (6th to 19th) related to
 309 more variable weather and low-pressure systems passing over the UK (Weather, 2018a;
 310 Weather, 2018b) (Fig. 5a). A transient decrease in O₃ is also apparent in the actual
 311 measured ozone in June 2018 (Supplementary Fig. S54).
 312



313
 314 Overall totals of annual associated hospital admissions are less than 1% different when
 315 using 2018 vs 2003 meteorology (-0.46% in Scotland to 0.91% in South West for 2030
 316 emissions, and 0.07% in Northern Ireland to 0.80% in South West for 2050 emissions).
 317 While the annual total changes are small between the two meteorological years (with the
 318 same emissions), short-term peaks and episodes of higher O₃ (and associated hospital
 319 admissions) are transiently influenced by the difference in weather patterns.

320

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 O₃ 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 O₃ concentrations are generally higher, the impact of emission reductions is to lower some of the peak O₃ values (Fig. 2b), and results in a reduction in respiratory hospital admissions associated with O₃ 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 O₃ 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).

<u>Emission year</u>	<u>2018</u>	<u>2030</u>	<u>2040</u>	<u>2050</u>
<u>O₃ exposure (µg m⁻³)</u>	<u>86.1</u>	<u>85.1</u>	<u>84.2</u>	<u>84.0</u>
<u>Mean attributable fraction</u>	<u>6.2%</u>	<u>6.2%</u>	<u>6.1%</u>	<u>6.1%</u>
<u>Total attributable hospital admissions over the 34-days (change from 2018)</u>	<u>4958</u>	<u>4904</u> <u>(-54)</u>	<u>4851</u> <u>(-107)</u>	<u>4841</u> <u>(-116)</u>
<u>Mean daily attributable hospital admissions (change from 2018)</u>	<u>146</u>	<u>144</u> <u>(-1.6)</u>	<u>143</u> <u>(-3.1)</u>	<u>142</u> <u>(-3.4)</u>

4 Discussion

Our results suggest that current policies driving anthropogenic emission changes will lead to an increase in future UK O₃ concentrations and hence in associated respiratory emergency

344 hospital admissions; for example, by 4.6% in 2050, an increase of 2,801 admissions across
345 the UK from 2018 assuming no population changes. Hospital admissions increase by 11.7%
346 by 2050 when population projections are also included (Table S3). Our previous work has
347 shown that in future there are likely to be improvements in UK air quality through reductions
348 in PM_{2.5} and NO₂, and a reduction in the mortality burden associated with long-term
349 exposure (Macintyre et al., 2023), whilst here we show that there may be a concurrent, albeit
350 smaller, increase in respiratory hospital admissions. When restricting the analysis to the
351 hottest days of the year, future anthropogenic emission reductions result in lower O₃
352 concentrations on hot days, and lower associated respiratory hospital admissions. While the
353 overall annual burden of associated respiratory hospital admissions increases (as hospital
354 admissions are generally greater in colder months, and the effect of emission reductions is
355 generally to increase O₃ concentrations in these months), on hot days O₃ concentrations are
356 lower in simulations with lower anthropogenic emissions, which could be beneficial for health
357 and care services during periods of hot weather (Table 3). The use of different
358 meteorological years in our analysis results in only very small (<1%) differences in the
359 annual total attributable daily hospital admissions, although there is a clear influence of
360 meteorology on day-to-day variation in O₃ concentrations and thus in associated hospital
361 admission (Fig. 5). It is not possible to know future meteorology exactly, but clearly there will
362 be some potential impact of interannual variability in meteorology on future O₃
363 concentrations and health burdens as well. As heatwaves (commonly associated with O₃
364 episodes) are projected to become more frequent and intense with climate change in the UK
365 (Murphy et al., 2018), there may be implications for response planning when periods of hot
366 weather are forecast, as there may also be health effects from increased O₃. While the aim
367 here is to examine the impact of changes in anthropogenic emissions, the role of natural
368 emissions of O₃ precursors (for example BVOC emissions that increase with temperature),
369 and hemispheric CH₄ may become more important in future as anthropogenic emissions
370 become smaller, and as temperatures rise. BVOC emissions are included in the model and
371 are dependent on the meteorology; though as the aim of the study is to evaluate the impacts
372 of anthropogenic air pollutant emission changes associated with current policies, we use the
373 same meteorological year to aid comparison. However, whilst BVOC emissions are the
374 same for simulations using the same meteorological year, the impact of the BVOC emissions
375 on O₃ formation will vary with the different anthropogenic VOC and NO_x emissions. Our
376 simulations extend to 2050 which has been shown to be about the point where future
377 temperature projections start to significantly diverge for different scenarios. It is not possible
378 to simulate future meteorology at the high spatial and temporal resolution, and over a large
379 enough spatial domain, required to drive BVOC emissions and other meteorologically-
380 dependent transboundary atmospheric chemistry processes within air quality models such

381 as the EMEP4UK model we use here. Instead, our selection of a relatively hot recent year
382 (2018) that may become a 50-50 likely summer by 2050 (as noted in the UKCP18 headline
383 findings report) will emulate potential representation of future conditions.

384
385 Regional variation in the impacts of emissions changes is generally modest; the largest
386 impacts of emissions changes are observed in London, which has the greatest reduction in
387 NO_x emissions in future (leading to enhanced O₃ through reduced titration with NO). At the
388 daily scale, the largest changes are to the lowest O₃ concentrations which are due to high
389 NO chemically depleting the O₃. By reducing NO emissions, the most polluted NO days are
390 reduced in future, leading to increases in O₃ on these days (Fig. 3). This effect has also
391 been observed during the sudden curtailment of emissions, particularly from transport,
392 during the 'lockdown' periods associated with COVID-19 measures in 2020 (AQEG, 2020;
393 Grange et al., 2021; Jephcote et al., 2021).

394
395 Our results here align with other studies assessing the impact of future emission changes on
396 O₃ concentrations which find similar increases near population centres, driven by NO_x
397 emission reductions (Heal et al., 2013; Hedegaard et al., 2013; Fenech et al., 2021). Very
398 few studies have quantified hospital admissions associated with O₃ across the UK (with the
399 majority examining mortality); our results for 2018 (60,488) are larger than those of Heal et
400 al. (2013, who reported 30,700 respiratory hospital admissions associated with O₃ exposure
401 in 2003, though our study has higher baseline admission rates, different background
402 emission scenarios and atmospheric model resolution, and a slightly larger ERF coefficient;
403 our study also has higher O₃ levels, with averages in the range 70-80 µg m⁻³ compared with
404 60-70 µg m⁻³. Stedman et al. (1997) estimate about 6% of respiratory emergency hospital
405 admissions are attributable to O₃, which compares with the figures in our study, noting this is
406 strongly affected by use or not of an O₃ concentration threshold, where health effects are
407 assumed to be attributed to the exposure only above a given threshold for effects, and no
408 health impacts attributed at concentrations below this. The choice of exposure coefficient
409 and concentration threshold for effects is important as this impacts the overall health burden
410 estimates, which are higher when no threshold is used (e.g. Stedman 1997 found a 20-fold
411 increase in burden estimate when using no threshold as opposed to a 50 ppbv threshold).
412 However, when assessing the impact of O₃ changes, the impact of the change appears
413 larger when a threshold is used; for example if a threshold for O₃ effects is assumed, health
414 burdens are more sensitive to changes in O₃ concentrations, although total health burdens
415 are roughly an order of magnitude lower (Stedman et al., 1997; Stedman and Kent, 2008;
416 Heal et al., 2013; COMEAP, 2022). Recent recommendations for the UK are not to use a
417 threshold for effects (COMEAP, 2022), which we followed in this work. There is some

418 evidence that temperature may act as a modifier for O₃ effects, and this may be important in
419 future if heatwaves and episodes of O₃ become more common or intense (Pattenden et al.,
420 2010; Atkinson et al., 2012). It is not possible to predict future changes in exposure-
421 response relationships. Changes in baseline hospital admissions may also influence overall
422 health impact estimates.

423

424 The UK has ~~an objective target~~ value of 100 µg m⁻³ for the daily maximum 8-hour mean O₃,
425 not to be exceeded more than 10 times a year as set out in the National Air Quality
426 Objectives as part of the Air Quality Standards Regulations 2010 (UK Government, 2010).

427 Our study shows that, in future, the number of days exceeding the target value of 100 µg m⁻³
428 is projected to increase in London (from 44 in 2018 to 57 days in 2050), North West (26 in
429 2018 to 36 in 2050), and East England (37 in 2018 to 48 in 2050), and decreases in others
430 (1 or 2 fewer days in Scotland, Northern Ireland and the South West), with small increases in
431 other areas. In 2021, the WHO updated its recommended air quality guideline (AQG) levels
432 for air pollutants, including O₃, with a new guideline of 100 µg m⁻³ for 8-hour mean O₃
433 (reduced from 120 µg m⁻³) not to be exceeded 3-4 times per year (World Health
434 Organization, 2021). Our analysis shows it may be harder for the UK to meet its objective
435 and the new WHO AQG values in future. We note however that the data presented above
436 are based on population-weighted mean values for the region or country and that changes in
437 daily maximum 8-hour mean O₃ at specific locations will vary.

438

439 At global scale, surface O₃ is expected to decrease slightly in future in low-NO_x regimes, due
440 to increased humidity leading to greater O₃ destruction, whereas in polluted regions, model
441 studies suggest an increase in O₃ (Szopa et al., 2021), with the influence of climate changes
442 on air quality being dependent on the particular emissions scenarios (Fiore et al., 2015).
443 Here we use fixed CH₄ and O₃ boundary conditions, and while wider hemispheric trends in
444 ozone from global trends in emissions of CH₄ will impact on UK concentrations, the direction
445 of change is unclear (AQEG, 2021).

446

447 ~~The results here and from our previous work~~ A previous study using the same modelling as
448 here (Macintyre et al., 2023) showed that ~~while~~ reducing emissions generally improves air
449 pollution (PM_{2.5} and NO₂) and the associated ~~health-mortality~~ burdens by around a third by
450 2050 (Macintyre et al., 2023), though as no threshold for effects is assumed, health impacts
451 still remain. Here however we show that, there can be some worsening effects related to
452 secondary pollutants such as O₃, though for hot days (that often coincide with higher O₃)
453 these may be reduced. While the overall impacts of emission reductions are beneficial for
454 health (Macintyre et al., 2023) (acknowledging that it is challenging to directly compare

455 mortality burdens and daily hospital admissions), we find that there may be increased effects
456 on emergency respiratory hospital admissions ~~particularly during air pollution episodes~~,
457 which is important for health system response planning when such episodes are forecast.
458 Our analysis in this study and a previous study (Macintyre et al., 2023) ~~both studies~~ provides
459 quantitative estimates of the potential effects to health of current policy commitments.
460 ~~Although overall health burden is simulated to decrease in future, because there are no~~
461 ~~thresholds for effects then health impacts associated with exposure to air pollution will~~
462 ~~remain in future.~~ The methods here use a simple burden calculation to estimate health
463 effects, though the specific impacts will depend on the speed and scale of emission
464 reductions. Our results indicate the likely impacts to health if emissions likely to be reached
465 at a particular indicative year in the future were to be met and sustained. While we examine
466 the impact of population size changes, potential changes in baseline hospital admission
467 rates are another source of uncertainty for future projections. An important point is that the
468 UK population is ageing; as the likelihood of living with chronic disease and susceptibility to
469 illness in general tends to increase with age, there may be increases in the baseline rates
470 of many health effects associated with air pollution. Future baseline rates of hospital
471 admissions ~~though these~~ are challenging to predict in any robust manner as many factors
472 influence these (e.g. circulating respiratory infections, rates of chronic respiratory disease,
473 influenza vaccine uptake and effectiveness) and thus are not considered here, which is a
474 limitation of the study. Another limitation is related to potential future changes in exposure-
475 response relationships; No known methodology exists to reliably predict these as these
476 depend on multiple factors such as the future air pollution composition, exposure estimates,
477 and population health, thus we do not examine this in the current analysis. Finally, the
478 effects of higher air pollution levels and air pollution episodes will undoubtedly be modified
479 by behaviours that affect exposure and inhaled dose, such as time spent indoors and
480 exercise undertaken.

481

482 **Conclusion**

483

484 The effect of future emission policy on air pollution in the UK has been modelled and a
485 quantitative HIA to estimate impacts on emergency respiratory hospital admissions ~~health~~
486 performed. While our previous work showed significant reductions in the mortality burden
487 due to lower PM_{2.5} and NO₂ concentrations, we estimate here that there will be a small
488 increase in O₃ exposure in future, and hence in associated hospital admissions, as a result
489 of lower NO in urban areas, where most people reside. The main impacts on O₃ occur on the
490 days with lowest O₃ (i.e. highest NO_x days) in regions that are already more polluted
491 generally, such as London. Emission control policy has clear benefits to health through

492 reducing PM_{2.5} and NO₂, though effects related to secondary pollutants such as O₃ can be
493 complex and should be acknowledged. As other air pollutants in the UK continue to decline
494 thanks to emission controls, it seems likely that the importance of O₃ will grow. Future work
495 on the health impacts of air pollution episodes in summer coinciding with heat should
496 consider potential worsening effects associated with local increasing O₃ concentrations, and
497 may need to be considered when ~~be a priority for better~~ planning ~~efor~~ public health
498 protection during these periods.

499

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518

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673

1 **Future impacts of O₃ on respiratory hospital admission in the UK from current**
2 **emissions policies**

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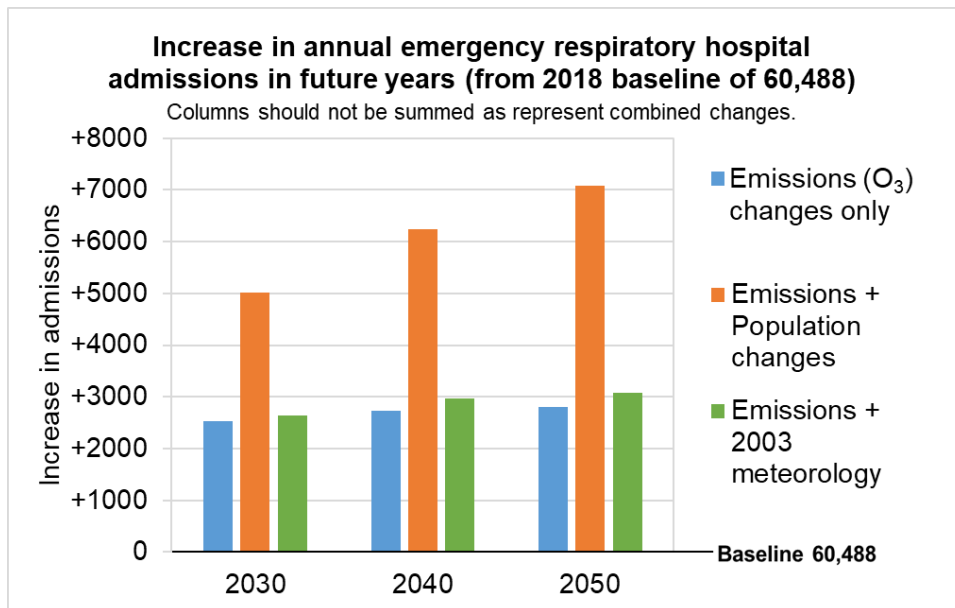
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18
19
20 **Highlights**

- 21 • Policies on anthropogenic emissions dominate health impacts of air pollution
- 22 • O₃ simulated at 3 km resolution for 2030, 2040 and 2050 emissions pathways
- 23 • Hospitalisations calculated with latest UK health impact assessment
24 recommendations
- 25 • 7078 increase in respiratory hospitalisations by 2050 for current policy c.f. 2018
- 26 • Estimated increases are 2-3 times lower if population growth is not included

27
28 **Graphical abstract**



29

30

31

32 **Keywords**

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

34

35 **Abstract**

36 Exposure to ambient ozone (O₃) is associated with impacts on human health. O₃ is a

37 secondary pollutant whose concentrations are determined inter alia by emissions of

38 precursors such as oxides of nitrogen (NO_x) and volatile organic compounds (VOCs), and

39 thus future health burdens depend on policies relating to climate and air quality. While

40 emission controls are expected to reduce levels of PM_{2.5} and NO₂ and their associated

41 mortality burdens, for secondary pollutants like O₃ the picture is less clear. Detailed

42 assessments are necessary to provide quantitative estimates of future impacts to support

43 decision-makers. We simulate future O₃ across the UK using a high spatial resolution

44 atmospheric chemistry model with current UK and European policy projections for 2030,

45 2040 and 2050, and use UK regional population-weighting and latest recommendations on

46 health impact assessment to quantify respiratory emergency hospital admissions associated

47 with short-term effects of O₃. We estimate 60,488 admissions in 2018, increasing by 4.2%,

48 4.5% and 4.6% by 2030, 2040 and 2050 respectively (assuming a fixed population).

49 Including future population growth, estimated emergency respiratory hospital admissions are

50 8.3%, 10.3% and 11.7% higher by 2030, 2040 and 2050 respectively. Increasing O₃

51 concentrations in future are driven by reduced nitric oxide (NO) in urban areas due to

52 reduced emissions, with increases in O₃ mainly occurring in areas with lowest O₃

53 concentrations currently. Meteorology influences episodes of O₃ on a day-to-day basis,

54 although a sensitivity study indicates that annual totals of hospital admissions are only
55 slightly impacted by meteorological year. While reducing emissions results in overall benefits
56 to population health (through reduced mortality due to long-term exposure to PM_{2.5} and
57 NO₂), due to the complex chemistry, as NO emissions reduce there are associated local
58 increases in O₃ close to population centres that may increase harms to health.

59

60

61 **1 Introduction & background**

62

63 Air pollution has detrimental impacts on human health. The main air pollutants associated
64 with impacts on human health are PM_{2.5} (particulate matter with aerodynamic diameter
65 <2.5 µm), nitrogen dioxide (NO₂), and ozone (O₃), with long-term exposure to ambient PM_{2.5}
66 and NO₂ in the UK estimated to have an effect equivalent to 29,000–43,000 deaths annually
67 (Mitsakou et al., 2022). Exposure to ambient O₃ is also associated with ill health, particularly
68 respiratory effects including exacerbation of asthma symptoms. O₃ is estimated to contribute
69 to over 5 million disability adjusted life years (DALYs) lost annually globally (Forouzanfar et
70 al., 2015). The evidence of O₃ impacts on health is strongest for associations between short-
71 term exposure (as quantified using the daily maximum 8-hour running mean O₃) and
72 respiratory and cardiovascular hospital admissions, though with more uncertainty for the
73 latter (COMEAP, 2015). Quantification of the impact on health of long-term exposure to O₃ is
74 not recommended by the UK's Committee on the Medical Effects of air Pollutants
75 (COMEAP) as there is currently insufficient evidence (COMEAP, 2015; Huangfu and
76 Atkinson, 2020; COMEAP, 2022).

77

78 Ozone is not directly emitted but is a secondary pollutant formed through complex
79 (photo)chemical reactions in the atmosphere involving NO_x (= NO + NO₂) and volatile
80 organic compound (VOC) emissions, including of methane (CH₄). The levels of methane
81 contribute to the longer-term (annual average) concentrations of O₃ that are also part of O₃
82 exposure and upon which the shorter-term variations in O₃ driven by non-methane VOCs are
83 superimposed. Concentrations of O₃ are also affected by meteorology, which influences its
84 build-up through photochemistry and air mass transport and dilution, and also via the rate of
85 O₃ deposition to the surface (Royal Society, 2008). High levels of NO_x (e.g. close to emission
86 sources) also act to limit O₃ formation, which is why reductions in NO_x emissions in urban
87 areas, largely associated with improvements in vehicle exhaust emission standards, have
88 led to modelled local increases in O₃ (Carnell et al., 2019).

89

90 Air pollution levels have generally improved in the UK over the past several decades, thanks
91 to environmental and public health initiatives. Carnell et al. (2019) reported that reductions in
92 air pollution levels between 1970 and 2010 led to a reduction in UK attributable mortality
93 associated with exposure to PM_{2.5} and NO₂ of 56% and 44% respectively; however, the
94 same study reported an increase in O₃-attributable respiratory mortality of 17%. This reflects
95 a different trend in response to recent air pollutant emissions changes for O₃ in urban areas,
96 where most people live, than for urban NO₂ and PM_{2.5} (AQEG, 2021). In particular, as noted
97 above, there is potential for local increases in O₃ due to reductions in emissions of NO_x, with
98 a recent report noting upwards trend in urban ozone of 5 – 9 µg m⁻³ between 2000-2019
99 (AQEG, 2021). A modelling study on possible UK net-zero policies showed that while longer-
100 term (April to September) mean O₃ concentrations may reduce, short-term O₃ metrics
101 revealed less O₃ suppression from reduced NO_x, particularly in winter (Williams et al., 2018).
102 The effect of reduced traffic NO_x emissions increasing urban O₃ was also illustrated during
103 the travel restrictions implemented as part of COVID-19 ‘lockdown’ restrictions (AQEG,
104 2020; Lee et al., 2020; Jephcote et al., 2021).

105
106 Changes in climate will also modify concentrations of surface O₃, as it will for other air
107 pollutants (von Schneidmesser et al., 2015; Doherty et al., 2017). However, the Doherty
108 et al. review concludes that for O₃ over Europe in the next few decades, changes in
109 anthropogenic emissions of the VOC and NO_x precursors will have much greater influence
110 on O₃ levels than changes in climate. For example, model simulations of impacts of climate
111 scenarios and compatible air pollutant emissions under CMIP5 scenarios showed O₃
112 exposure metrics over Europe in 2050 change by 3% (±8%) to 5% (±11%) due to changes in
113 climate, but by -24% (±10%) to -43% (±7%) due to emission changes (Colette et al., 2013).
114 Over the UK, changes in annual mean O₃ by 2030 (compared to 2003) due to changes in
115 precursor emissions (-3.0 to +3.5 ppbv) are larger than for a +5°C scenario (+1.0 to
116 +1.5 ppbv) including temperature effects on biogenic VOCs, chemical reaction rates and O₃
117 dry deposition, with the latter being comparable to the changes in O₃ arising from normal
118 interannual meteorological variability (Heal et al., 2013). Resulting changes in mortality
119 burdens were 4.1% associated with O₃ changes due to +5°C, whereas they were 15.6% to
120 27.7% for emissions changes (depending on emission scenario), assuming no threshold for
121 effects (Heal et al., 2013).

122
123 In a previous study we performed an up-to-date quantitative impact assessment of the
124 mortality burdens associated with long-term effects of PM_{2.5} and NO₂ in 2030, 2040 and
125 2050 using a high spatial resolution atmospheric chemistry and aerosol model with the latest
126 official UK and European emissions datasets (Macintyre et al., 2023). We found that

127 mortality burdens associated with these two pollutants were reduced by over 30% beyond
128 2030. As O₃ is also impacted by such emission policies, but in more complex ways as
129 described above, we expand our analysis here to consider impacts on respiratory hospital
130 admissions associated with short-term exposure to O₃, using a similar health impact
131 assessment (HIA), to more fully capture future impacts to health associated with changing
132 emissions.

133
134

135 **2 Methods**

136

137 ***2.1 Air pollution modelling and emissions***

138 Simulations of UK present-day and future air quality at 3 km × 3 km spatial resolution and
139 hourly temporal resolution were undertaken using the EMEP4UK¹ atmospheric chemistry
140 transport model version rv4.36. The chemical model was driven by hourly meteorology from
141 the Weather Research and Forecasting (WRF) model version 4.2.2 (wrf-model.org). The
142 EMEP4UK-WRF modelling system has been widely used to simulate historic, present-day
143 and potential future air quality over the UK (Heal et al., 2013; Vieno et al., 2014; Vieno et al.,
144 2016; Nemitz et al., 2020), including provision of evidence on air quality to the UK
145 government (HPA, 2012; AQEG, 2013; AQEG, 2017; AQEG, 2021). Details of the model
146 system have been described previously in our companion HIA study (Macintyre et al., 2023)
147 and in the above references, and are also detailed in the Supplementary Material, so only a
148 brief description of the model runs used in this study is provided here.

149

150 Anthropogenic emissions of NO_x, NH₃, SO₂, CO, NMVOC (non-methane VOC), PM_{2.5} and
151 PM_{co} (coarse particulate matter, aerodynamic diameter >2.5 µm and <10 µm) for the 2018
152 simulation for the UK were taken from the National Atmospheric Emissions Inventory
153 (naei.beis.gov.uk). UK emissions for model simulations of atmospheric composition in 2030,
154 2040 and 2050 use the 'business as usual (BAU)', also referred to as 'baseline' scenarios
155 developed for Defra (ApSimon et al., 2019; Defra, 2022; Oxley et al., 2023) which reflect
156 assumed trends for anthropogenic emissions under existing interventions and policies
157 relating to air quality, including adjustments to include the projected impact of recent policy
158 not included in the 2018 NAEI projections. Emissions projections for the rest of the European
159 domain use official anthropogenic EMEP emissions fields for the corresponding years
160 (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).

161 evaluate the impact of changes in anthropogenic emissions on changes in health burden.
 162 Biogenic VOC (BVOC) emissions are included in the model and are dependent on the
 163 meteorological year (here held constant to evaluate changes in anthropogenic emissions),
 164 and thus will be the same in simulations using the same meteorological year. Additionally, a
 165 sensitivity experiment was run for future years using 2003 meteorology, as an example of a
 166 hot year in the UK for comparison. The CH₄ concentration and the O₃ concentrations at the
 167 boundary of the extended European domain were fixed at 2018 values.

168

169 **2.2 Exposure metric and population-weighting**

170 The daily maximum of the 8-hour running mean O₃ concentration was calculated from
 171 gridded hourly model output for each of the simulation years at the native grid resolution, to
 172 match the O₃ exposure metric used in the epidemiology (see equations in Section 2.3). This
 173 O₃ exposure metric was population-weighted to create regional estimates for the nine
 174 regions in England, and for Scotland, Wales and Northern Ireland (Fig. 1b) using 100 m
 175 gridded residential population information for England, Scotland and Wales (National
 176 Population Database, 2020), and at 1 km for Northern Ireland (Reis et al., 2017).

177

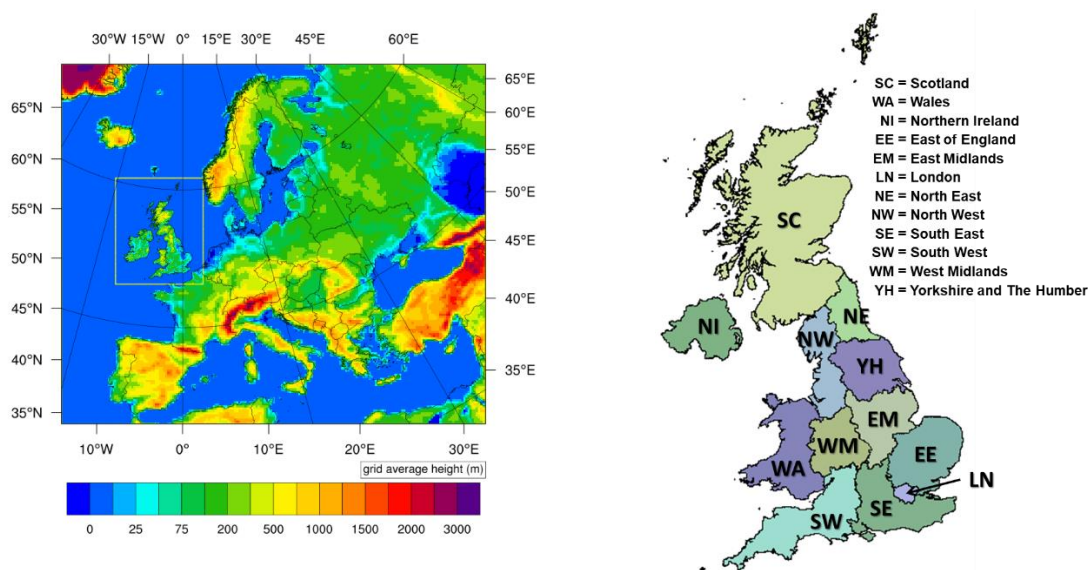


Figure 1. (a) The EMEP4UK model domain showing the 3 km x 3 km resolution British Isles domain (yellow box) nested within an extended Europe domain at 27 km x 27 km (plotted variable is altitude). (b) Regions used in the health impact assessment analysis.

178

179 **2.3 Health impact calculations**

180 The hospital admissions associated with short-term exposure to O₃ (H_{aq}) were estimated as
 181 follows: $H_{aq} = H_T \times AF$, where H_T is the total emergency respiratory hospital admissions (all-
 182 ages) in the region, and AF is attributable fraction of the health outcome in each region
 183 associated with exposure to O₃. AF is calculated based on the percent increase in

184 emergency respiratory hospital admissions per 10 $\mu\text{g m}^{-3}$ O_3 reported from meta-analyses of
185 time-series studies. We used an exposure-response coefficient corresponding to an increase
186 in emergency respiratory hospital admissions (all-ages) of 0.75% (95% CI: 0.30%,1.2%) per
187 10 $\mu\text{g m}^{-3}$ daily maximum 8-hour running mean O_3 with no threshold for effects (COMEAP,
188 2015; COMEAP, 2022).

189
190 Daily emergency hospital admissions for respiratory causes (all-ages) were obtained from
191 ONS for the nine regions in England (extracted via UKHSA DataLake), from Public Health
192 Scotland for Scotland, and provided by Northern Ireland Statistical and Research Agency
193 (NISRA) for Northern Ireland for 2018. For Wales, daily data was not available so annual
194 totals for respiratory causes and emergency admission type were obtained from Patient
195 Episode Database for Wales (PEDW, year 2018/19 dhw.nhs.wales), and distributed across
196 days in the year using the daily cycle of the data for England. Annual totals are shown in
197 Supplementary Table S1.

198
199 To account for future changes in population, the HIA results were normalised with mid-year
200 population estimates for 2018 for each region, and then scaled for future years using
201 population projections. ONS produce a range of population projections for the UK based on
202 assumptions about future fertility, mortality, and migration trends (based on long-term
203 demographic trends), with different scenarios ('variant' projections) produced based on
204 high/low fertility, migration, and mortality. We used the 'principal' population projections
205 representing a middle estimate for future population demographics (interim 2020-based,
206 (ONS, 2022)²). Projections are available for Scotland, Wales, Northern Ireland, and England
207 as a whole, so regional totals in England are scaled uniformly. Population totals are shown in
208 Supplementary Table S2.

209
210

211 **3 Results**

212

213 ***3.1 Impact of future emissions on O_3 and associated hospital admissions***

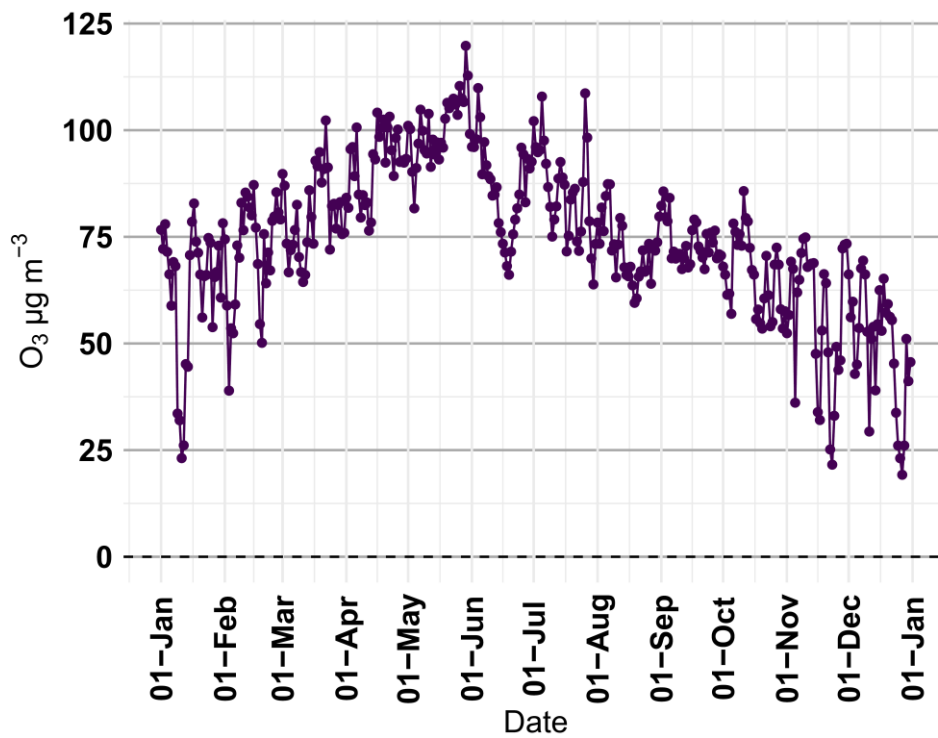
214 Across the UK, population-weighted O_3 is generally highest in the spring and early summer
215 (April to June) and lower in winter, with some very low values occurring in colder months
216 (November to February, Fig. 2a). This seasonal cycle is consistent with surface observations

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.

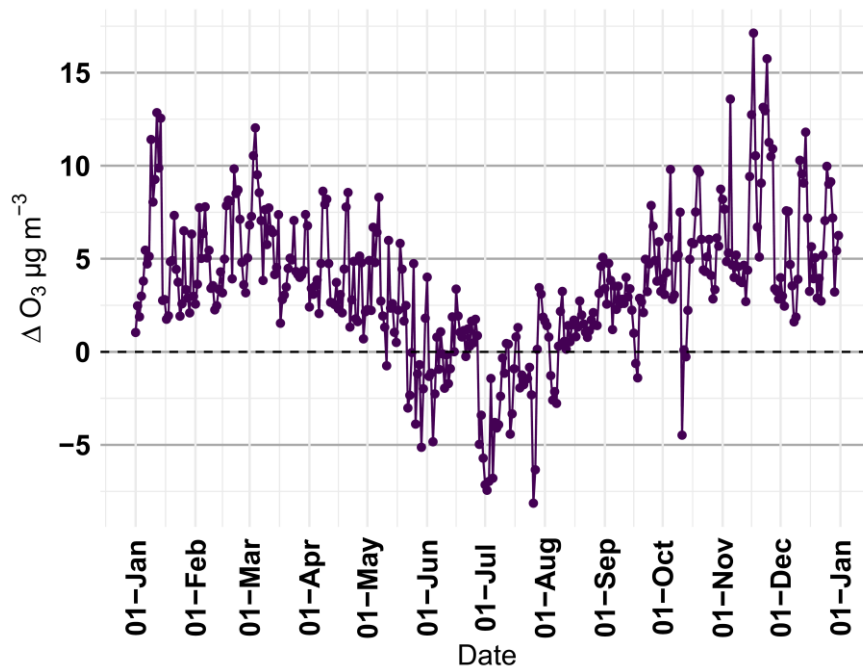
217 of UK O₃ (AQEG, 2021). The lowest values appear mostly in the London region, with the
218 West Midlands, East of England, and North west also showing low excursions (Fig. 3).
219 Comparing the impact of 2050 emissions against 2018 emissions shows an overall increase
220 in O₃ (Fig. 2b,c). The increases in O₃ under 2050 emissions are in areas where O₃ is lowest
221 with 2018 emissions, thus the greatest increases in O₃ 2050 are in those regions showing
222 lower values in 2018 (Fig. 3). In winter months (November to March) the O₃ concentrations
223 increase in response to emission changes across all regions, whereas for warmer months
224 the changes are mixed, with some increases and some decreases (Fig 2b).

225
226
227

(a) 2018 O₃ exposure (population-weighted value for the UK as a whole)



(b) Absolute difference in 2050 O₃ exposure with respect to 2018 O₃ exposure (i.e., positive indicates higher O₃ with 2050 emissions as compared with 2018)



(c) Relative difference in 2050 O₃ exposure with respect to 2018 O₃ exposure

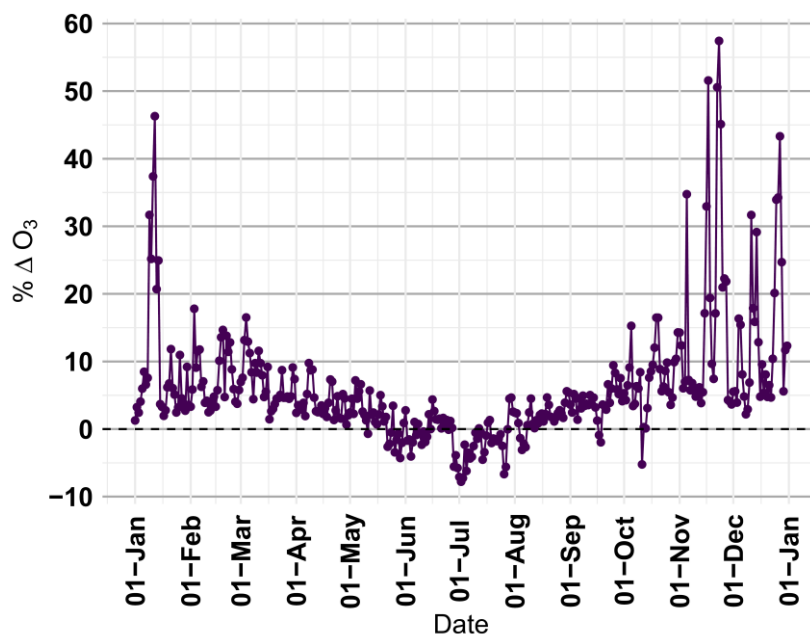


Figure 2: (a) UK population-weighted O₃ exposure (daily maximum 8-hour running mean) in 2018; (b) Difference in 2050 compared to 2018; (c) percent difference in 2050 compared to 2018.

228

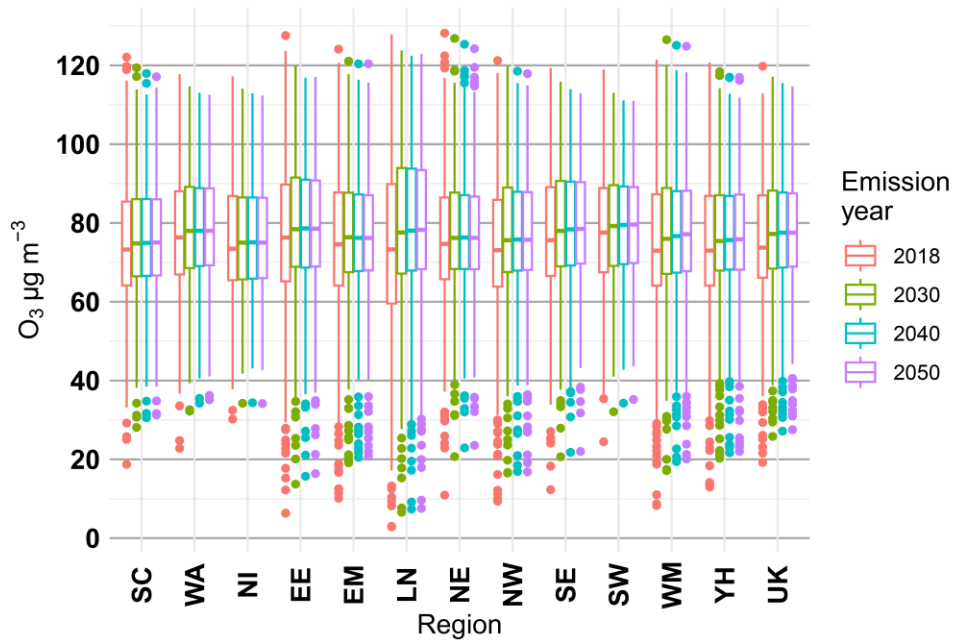
229 Examining the overall annual distribution of O₃ exposure by region and emission years
 230 (Fig. 3) reveals small increases in median daily O₃ exposure, of around 2 – 4 μg m⁻³, and up
 231 to 6 μg m⁻³ in London, with some individual days showing >20 μg m⁻³ increase (Fig. 3b).

232 There is a narrowing of the range of exposures by 2050 (Fig. 3a), as high values in summer
 233 are slightly reduced, and low values in winter months are increased (Fig. 2b). While the

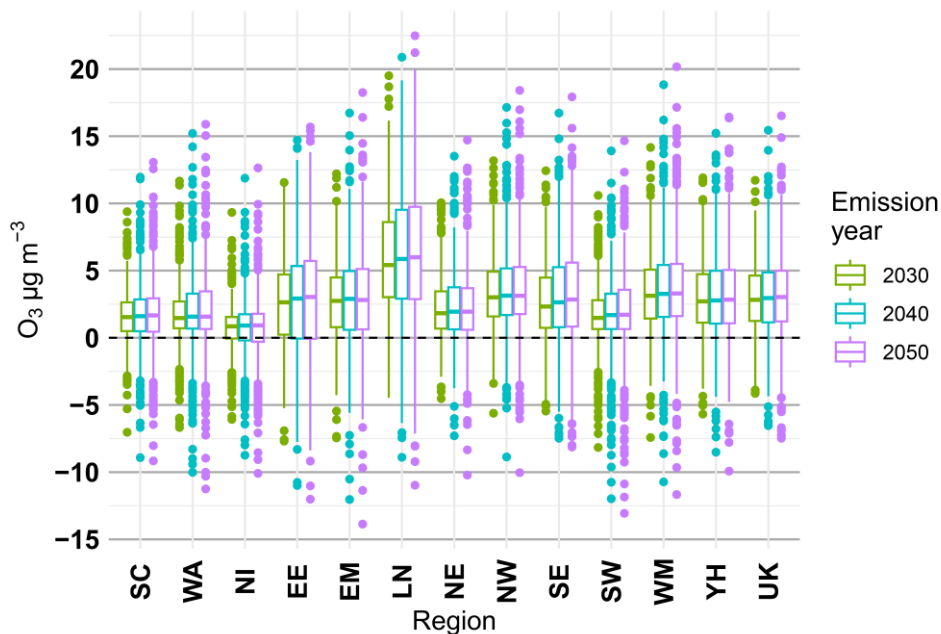
234 range of exposure narrows in future, the range in the difference compared to 2018 appears

235 to expand (Fig. 3b) indicating less variations in the daily concentrations; this is due to the
236 decreases in O₃ occurring where there is existing higher O₃ (so peak O₃ values in summer
237 are reduced), and the increases in O₃ being where low O₃ values currently occur, particularly
238 so for the instances of very low O₃ values. As the increases in O₃ (due to less suppression
239 by NO) occur on background values that are already small, this leads to very high apparent
240 changes when expressed as a percentage (Fig. 3c).
241

(a) O₃ exposure



(b) absolute differences in O₃ exposure compared to 2018



(c) relative differences in O₃ exposure compared to 2018

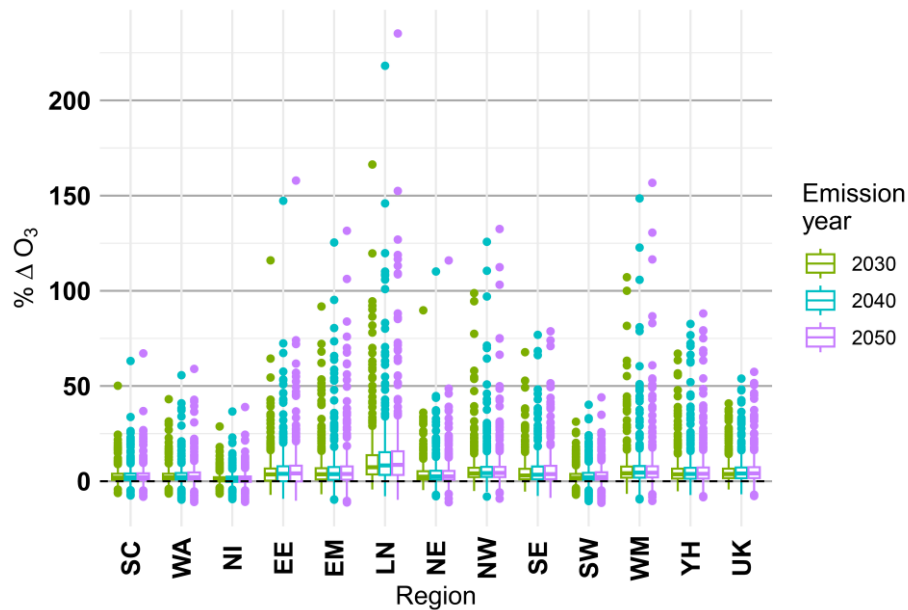


Figure 3: Boxplots* of daily population-weighted O₃ exposure (based on daily max 8-hour running mean) across all regions and different emission years (meteorology is for 2018 in all simulations). (a) Daily absolute values; (b) daily absolute differences compared to 2018; (c) percent differences compared 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.

242

243 The annual mean daily fraction of emergency respiratory hospital admissions attributable to
244 short-term effects of O₃ exposure was 5.4% in 2018 (with mean ranging from 5.25% in
245 London to 5.65% in the South West), rising slightly to 5.6% by 2050 (range 5.52% in
246 Scotland to 5.78% in the South West) (Fig. 4). The annual cycle of the attributable fractions
247 is in Supplementary Fig. S1. While attributable fractions are driven by air pollution
248 concentrations, the absolute totals of attributable hospital admissions are also influenced by
249 baseline admission rates and population in each region. Northern Ireland, Scotland, and the
250 North West have higher baseline admission rates, and the South East, London and the
251 North West have higher populations, leading to the highest overall attributable hospital
252 admissions being estimated in Scotland, North West and South East (Fig. 4). Given that the
253 changes in future O₃ concentration are comparatively small, the differences in annual
254 attributable fractions and hospital admissions in future years compared with 2018 shown in
255 Fig. 4 are likewise relatively modest. (The values of attributable fractions and hospital
256 admissions using the confidence interval range on the exposure response function are
257 shown in Supplementary Fig. S2.)

258

259 Our analysis suggests that for 2018 the annual total of daily emergency hospital admissions
 260 for respiratory causes across the UK associated with short-term exposure to O₃ was 60,488
 261 (95% CI: 24,673 – 94,927), and that this will be higher by 4.2%, 4.5%, and 4.6% in 2030,
 262 2040 and 2050 respectively (relative to 2018) reaching 63,289 (95% CI: 25,827 – 99,278)
 263 with 2050 emissions, an increase of 2,801 (1,154 – 4,352) (Table 1). The overall estimated
 264 associated hospital admissions in each region are also influenced by baseline admission
 265 rates and population size in each region, with the North West, London and South East
 266 having larger populations, and Scotland having larger admission rates.
 267

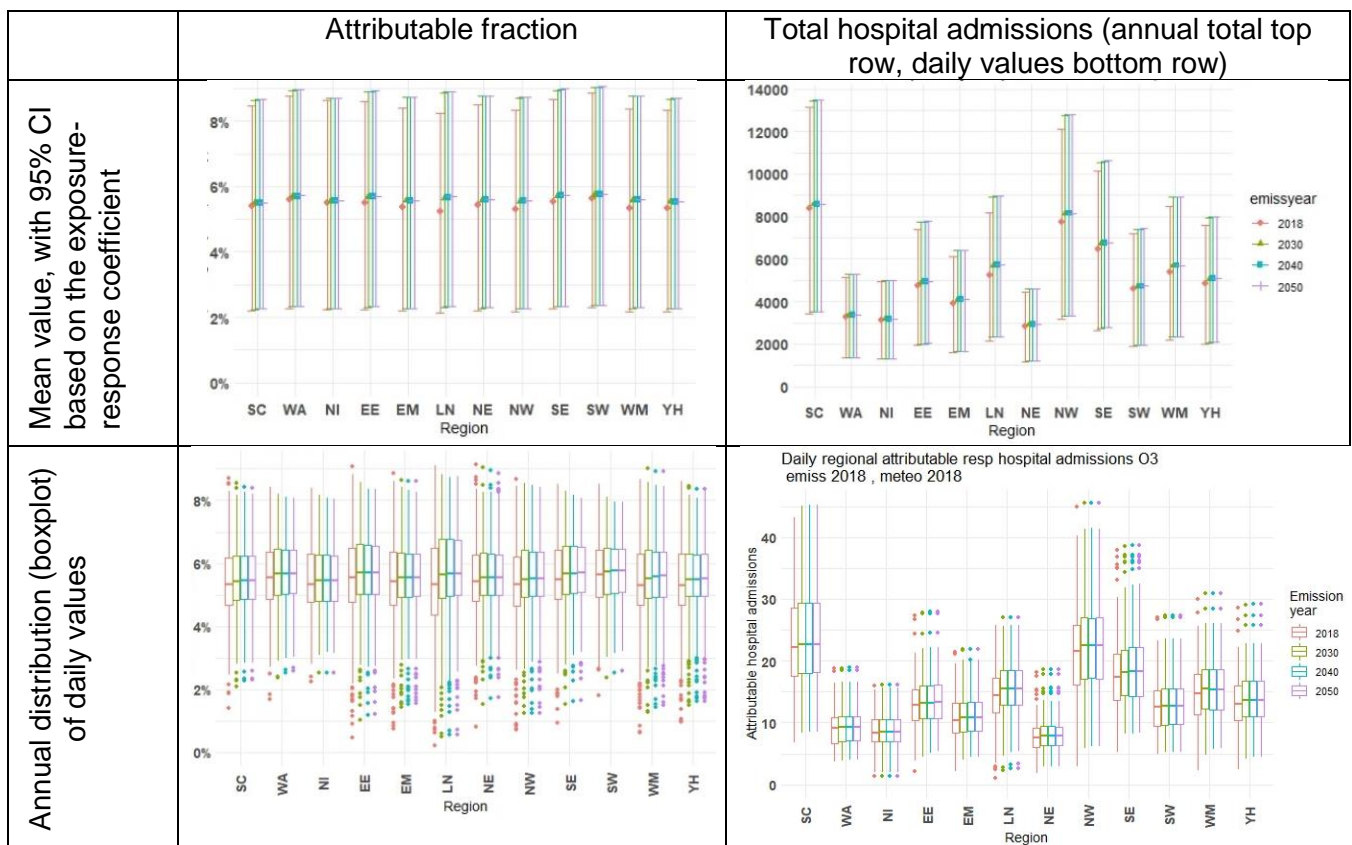


Figure 4: Regional attributable fraction (left column) and annual total respiratory emergency hospital admissions (right column) associated with short-term exposure to O₃. The top row shows annual totals, with error bars showing the 90% confidence interval on the exposure-response coefficient. The bottom 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.)

268

269

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 (25,8270 – 99,278)

270

271 When accounting for changes in population size in the future (scaled based on the
272 population changes in Table S2), the annual total of daily hospital admissions for the UK in
273 2050 rises from 63,289 to 67,566 (+4,277) (Table 2), meaning that the increases from the
274 2018 baseline of 4.2%, 4.5%, and 4.6% in 2030, 2040 and 2050 respectively (due to O₃
275 changes only), are now increased to 8.3%, 10.3% and 11.7% respectively (due to both O₃
276 and population changes) (full breakdown of results in Table S3). This total hides some
277 regional variation, as although overall UK population increases by 2050, the population in
278 Scotland is expected to rise in 2030, but then fall thereafter and by 2050 is 1.9% lower than
279 its 2018 value (Table S2). The increase in hospital admissions in Scotland (due to O₃ only
280 with no population change) of 222 by 2050, becomes an increase of 55 hospital admissions
281 over the same period when accounting for the shrinking population. In Northern Ireland,
282 population is projected to increase in 2030 and 2040, but to reduce in 2050 closer to 2030
283 levels (Table S2). In all other regions, population changes increase the health burden, with
284 the greatest combined (O₃ and population) changes in hospital admissions by 2050 being in
285 London (+19.1%), the West Midlands (+14.9%), and North West (+14.7%) with respect to
286 the 2018 baseline.

287

Table 2: Respiratory emergency hospital admissions associated with O₃ 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

288

289

290 **3.2 Impact of inter-annual variation in meteorology on O₃ and hospital admissions**

291 The future emission years were also run with 2003 meteorology as another historical ‘hot’
292 year for the UK as a sensitivity study for the impact of interannual meteorology on O₃
293 concentrations. While the annual means of future population-weighted daily maximum 8-
294 hour mean concentrations are very similar when using either 2018 or 2003 meteorology
295 (less than 1 µg m⁻³ difference on a UK mean of all such values of 78 µg m⁻³), there are
296 differences in the annual cycles of the daily concentrations, as illustrated in Fig. 5 for 2050
297 emissions. Overall, the annual cycles both show higher concentrations in April and May (as
298 expected), with some peaks during heatwaves and anticyclonic airflows; for example the
299 effects on simulated O₃ of using the meteorology associated with the European heatwave in
300 early August 2003³ are clearly visible (Fig. 5b). There is also a particular O₃ peak associated
301 with the 16 April 2003 meteorology, coinciding with an unusually hot day for that time of year
302 (26.7°C was recorded in Northolt, Greater London) with clear skies and south easterly air
303 flow, with population-weighted daily maximum 8-hour mean O₃ exceeding 120 µg m⁻³ in
304 West Midlands, North West, East Midlands, and Yorkshire and The Humber (Supplementary
305 Fig. S4). May 2018 was the sunniest May on record, leading to high O₃ values when using
306 that month’s meteorology, followed by a decline in O₃ in early June (6th to 19th) related to
307 more variable weather and low-pressure systems passing over the UK (Weather, 2018a;
308 Weather, 2018b) (Fig. 5a). A transient decrease in O₃ is also apparent in the actual
309 measured ozone in June 2018 (Supplementary Fig. S5).

310

(a)	(b)
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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.

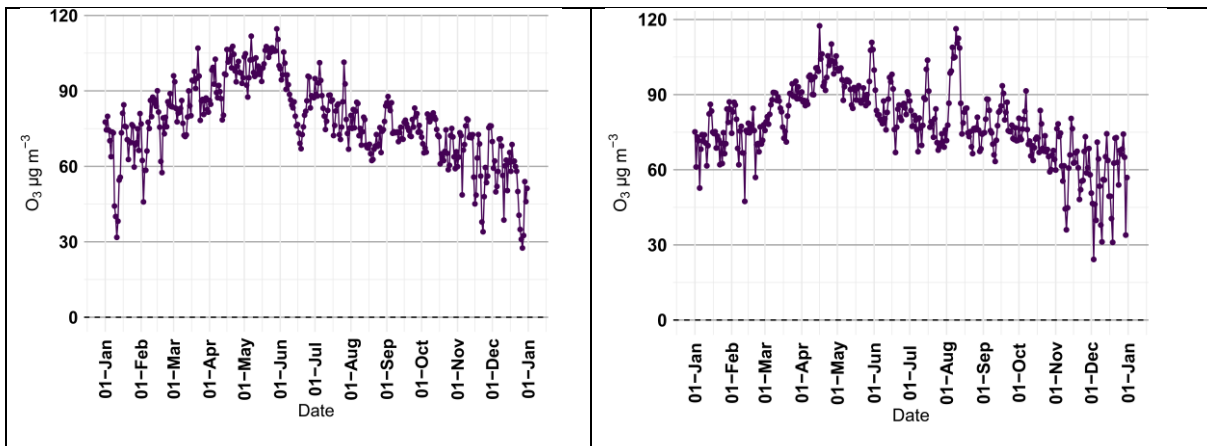


Figure 5: Population-weighted UK O₃ concentrations for 2050 emissions using (a) 2018 meteorology and (b) 2003 meteorology.

311

312 Overall totals of annual associated hospital admissions are less than 1% different when
 313 using 2018 vs 2003 meteorology (−0.46% in Scotland to 0.91% in South West for 2030
 314 emissions, and 0.07% in Northern Ireland to 0.80% in South West for 2050 emissions).
 315 While the annual total changes are small between the two meteorological years (with the
 316 same emissions), short-term peaks and episodes of higher O₃ (and associated hospital
 317 admissions) are transiently influenced by the difference in weather patterns.

318

319 **3.3 Impact of emissions changes on hot days**

320 As periods of hot weather and O₃ episodes often overlap, we also present the impact of
 321 emissions changes on respiratory hospital admissions with the analysis restricted to days
 322 with higher temperatures. To define a ‘hot day’, studies typically consider different
 323 percentiles of the annual distribution of daily mean temperatures, and we selected here the
 324 95th percentile to define a ‘hot day’ (as also used by Pattenden et al. (2010)). Taking the
 325 Central England Temperature (CET) as an indicator for UK temperature (Parker et al.,
 326 1992), gives 18.4°C as the 95th percentile of the annual distribution of daily mean
 327 temperature (with a total of 34 days above this temperature in 2018). During summer when
 328 O₃ concentrations are generally higher, the impact of emission reductions is to lower some
 329 of the peak O₃ values (Fig. 2b), and results in a reduction in respiratory hospital admissions
 330 associated with O₃ exposure for these days (Table 3). This would indicate that while the
 331 overall annual total of respiratory emergency hospital admissions across the year increases
 332 in response to anthropogenic emissions changes (Fig. 4, Table 1), those admissions
 333 occurring on the hottest days (when O₃ concentrations are highest) are actually reduced,
 334 demonstrating a benefit of anthropogenic emission controls on hot days.

335

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).

	2018	2030	2040	2050
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)

336

337

338 4 Discussion

339

340 Our results suggest that current policies driving anthropogenic emission changes will lead to
 341 an increase in future UK O₃ concentrations and hence in associated respiratory emergency
 342 hospital admissions; for example, by 4.6% in 2050, an increase of 2,801 admissions across
 343 the UK from 2018 assuming no population changes. Hospital admissions increase by 11.7%
 344 by 2050 when population projections are also included (Table S3). Previous work has shown
 345 that in future there are likely to be improvements in UK air quality through reductions in PM_{2.5}
 346 and NO₂, and a reduction in the mortality burden associated with long-term exposure
 347 (Macintyre et al., 2023), whilst here we show that there may be a concurrent, albeit smaller,
 348 increase in respiratory hospital admissions. When restricting the analysis to the hottest days
 349 of the year, future anthropogenic emission reductions result in lower O₃ concentrations on
 350 hot days, and lower associated respiratory hospital admissions. While the overall annual
 351 burden of associated respiratory hospital admissions increases (as hospital admissions are
 352 generally greater in colder months, and the effect of emission reductions is generally to
 353 increase O₃ concentrations in these months), on hot days O₃ concentrations are lower in
 354 simulations with lower anthropogenic emissions, which could be beneficial for health and
 355 care services during periods of hot weather (Table 3). The use of different meteorological
 356 years in our analysis results in only very small (<1%) differences in the annual total
 357 attributable daily hospital admissions, although there is a clear influence of meteorology on
 358 day-to-day variation in O₃ concentrations and thus in associated hospital admission (Fig. 5).

359 It is not possible to know future meteorology exactly, but clearly there will be some potential
360 impact of interannual variability in meteorology on future O₃ concentrations and health
361 burdens as well. As heatwaves (commonly associated with O₃ episodes) are projected to
362 become more frequent and intense with climate change in the UK (Murphy et al., 2018),
363 there may be implications for response planning when periods of hot weather are forecast,
364 as there may also be health effects from increased O₃. While the aim here is to examine the
365 impact of changes in anthropogenic emissions, the role of natural emissions of O₃
366 precursors (for example BVOC emissions that increase with temperature), and hemispheric
367 CH₄ may become more important in future as anthropogenic emissions become smaller, and
368 as temperatures rise. BVOC emissions are included in the model and are dependent on the
369 meteorology; though as the aim of the study is to evaluate the impacts of anthropogenic air
370 pollutant emission changes associated with current policies, we use the same
371 meteorological year to aid comparison. However, whilst BVOC emissions are the same for
372 simulations using the same meteorological year, the impact of the BVOC emissions on O₃
373 formation will vary with the different anthropogenic VOC and NO_x emissions. Our simulations
374 extend to 2050 which has been shown to be about the point where future temperature
375 projections start to significantly diverge for different scenarios. It is not possible to simulate
376 future meteorology at the high spatial and temporal resolution, and over a large enough
377 spatial domain, required to drive BVOC emissions and other meteorologically-dependent
378 transboundary atmospheric chemistry processes within air quality models such as the
379 EMEP4UK model we use here. Instead, our selection of a relatively hot recent year (2018)
380 that may become a 50-50 likely summer by 2050 (as noted in the UKCP18 headline findings
381 report) will emulate potential representation of future conditions.

382

383 Regional variation in the impacts of emissions changes is generally modest; the largest
384 impacts of emissions changes are observed in London, which has the greatest reduction in
385 NO_x emissions in future (leading to enhanced O₃ through reduced titration with NO). At the
386 daily scale, the largest changes are to the lowest O₃ concentrations which are due to high
387 NO chemically depleting the O₃. By reducing NO emissions, the most polluted NO days are
388 reduced in future, leading to increases in O₃ on these days (Fig. 3). This effect has also
389 been observed during the sudden curtailment of emissions, particularly from transport,
390 during the 'lockdown' periods associated with COVID-19 measures in 2020 (AQEG, 2020;
391 Grange et al., 2021; Jephcote et al., 2021).

392

393 Our results here align with other studies assessing the impact of future emission changes on
394 O₃ concentrations which find similar increases near population centres, driven by NO_x
395 emission reductions (Heal et al., 2013; Hedegaard et al., 2013; Fenech et al., 2021). Very

396 few studies have quantified hospital admissions associated with O₃ across the UK (with the
397 majority examining mortality); our results for 2018 (60,488) are larger than those of Heal et
398 al. (2013, who reported 30,700 respiratory hospital admissions associated with O₃ exposure
399 in 2003, though our study has higher baseline admission rates, different background
400 emission scenarios and atmospheric model resolution, and a slightly larger ERF coefficient;
401 our study also has higher O₃ levels, with averages in the range 70-80 µg m⁻³ compared with
402 60-70 µg m⁻³. Stedman et al. (1997) estimate about 6% of respiratory emergency hospital
403 admissions are attributable to O₃, which compares with the figures in our study, noting this is
404 strongly affected by use or not of an O₃ concentration threshold, where health effects are
405 assumed to be attributed to the exposure only above a given threshold for effects, and no
406 health impacts attributed at concentrations below this. The choice of exposure coefficient
407 and concentration threshold for effects is important as this impacts the overall health burden
408 estimates, which are higher when no threshold is used (e.g. Stedman 1997 found a 20-fold
409 increase in burden estimate when using no threshold as opposed to a 50 ppbv threshold).
410 However, when assessing the impact of O₃ changes, the impact of the change appears
411 larger when a threshold is used; for example if a threshold for O₃ effects is assumed, health
412 burdens are more sensitive to changes in O₃ concentrations, although total health burdens
413 are roughly an order of magnitude lower (Stedman et al., 1997; Stedman and Kent, 2008;
414 Heal et al., 2013; COMEAP, 2022). Recent recommendations for the UK are not to use a
415 threshold for effects (COMEAP, 2022), which we followed in this work. There is some
416 evidence that temperature may act as a modifier for O₃ effects, and this may be important in
417 future if heatwaves and episodes of O₃ become more common or intense (Pattenden et al.,
418 2010; Atkinson et al., 2012). It is not possible to predict future changes in exposure-
419 response relationships. Changes in baseline hospital admissions may also influence overall
420 health impact estimates.

421
422 The UK has an objective value of 100 µg m⁻³ for the daily maximum 8-hour mean O₃, not to
423 be exceeded more than 10 times a year as set out in the National Air Quality Objectives as
424 part of the Air Quality Standards Regulations 2010 (UK Government, 2010). Our study
425 shows that, in future, the number of days exceeding the target value of 100 µg m⁻³ is
426 projected to increase in London (from 44 in 2018 to 57 days in 2050), North West (26 in
427 2018 to 36 in 2050), and East England (37 in 2018 to 48 in 2050), and decreases in others
428 (1 or 2 fewer days in Scotland, Northern Ireland and the South West), with small increases in
429 other areas. In 2021, the WHO updated its recommended air quality guideline (AQG) levels
430 for air pollutants, including O₃, with a new guideline of 100 µg m⁻³ for 8-hour mean O₃
431 (reduced from 120 µg m⁻³) not to be exceeded 3-4 times per year (World Health
432 Organization, 2021). Our analysis shows it may be harder for the UK to meet its objective

433 and the new WHO AQG values in future. We note however that the data presented above
434 are based on population-weighted mean values for the region or country and that changes in
435 daily maximum 8-hour mean O₃ at specific locations will vary.

436

437 At global scale, surface O₃ is expected to decrease slightly in future in low-NO_x regimes, due
438 to increased humidity leading to greater O₃ destruction, whereas in polluted regions, model
439 studies suggest an increase in O₃ (Szopa et al., 2021), with the influence of climate changes
440 on air quality being dependent on the particular emissions scenarios (Fiore et al., 2015).

441 Here we use fixed CH₄ and O₃ boundary conditions, and while wider hemispheric trends in
442 ozone from global trends in emissions of CH₄ will impact on UK concentrations, the direction
443 of change is unclear (AQEG, 2021).

444

445 A previous study using the same modelling as here showed that reducing emissions
446 generally improves air pollution (PM_{2.5} and NO₂) and the associated mortality burdens by
447 around a third by 2050 (Macintyre et al., 2023), though as no threshold for effects is
448 assumed, health impacts still remain. Here however we show that there can be some
449 worsening effects related to secondary pollutants such as O₃, though for hot days (that often
450 coincide with higher O₃) these may be reduced. While the overall impacts of emission
451 reductions are beneficial for health (Macintyre et al., 2023) (acknowledging that it is
452 challenging to directly compare mortality burdens and daily hospital admissions), we find that
453 there may be increased effects on emergency respiratory hospital admissions, which is
454 important for health system response planning when such episodes are forecast. Our
455 analysis in this study and a previous study (Macintyre et al., 2023) provides quantitative
456 estimates of the potential effects to health of current policy commitments. The methods here
457 use a simple burden calculation to estimate health effects, though the specific impacts will
458 depend on the speed and scale of emission reductions. Our results indicate the likely
459 impacts to health if emissions likely to be reached at a particular indicative year in the future
460 were to be met and sustained. While we examine the impact of population size changes,
461 potential changes in baseline hospital admission rates are another source of uncertainty for
462 future projections. An important point is that the UK population is ageing; as the likelihood of
463 living with chronic disease and susceptibility to illness in general tends to increase with age,
464 there may be increases in the baseline rates of many health effects associated with air
465 pollution. Future baseline rates of hospital admissions are challenging to predict in any
466 robust manner as many factors influence these (e.g. circulating respiratory infections, rates
467 of chronic respiratory disease, influenza vaccine uptake and effectiveness) and thus are not
468 considered here, which is a limitation of the study. Another limitation is related to potential
469 future changes in exposure-response relationships; No known methodology exists to reliably

470 predict these as these depend on multiple factors such as the future air pollution
471 composition, exposure estimates, and population health, thus we do not examine this in the
472 current analysis. Finally, the effects of higher air pollution levels and air pollution episodes
473 will undoubtedly be modified by behaviours that affect exposure and inhaled dose, such as
474 time spent indoors and exercise undertaken.

475

476 **Conclusion**

477

478 The effect of future emission policy on air pollution in the UK has been modelled and a
479 quantitative HIA to estimate impacts on emergency respiratory hospital admissions
480 performed. While our previous work showed significant reductions in the mortality burden
481 due to lower PM_{2.5} and NO₂ concentrations, we estimate here that there will be a small
482 increase in O₃ exposure in future, and hence in associated hospital admissions, as a result
483 of lower NO in urban areas, where most people reside. The main impacts on O₃ occur on the
484 days with lowest O₃ (i.e. highest NO_x days) in regions that are already more polluted
485 generally, such as London. Emission control policy has clear benefits to health through
486 reducing PM_{2.5} and NO₂, though effects related to secondary pollutants such as O₃ can be
487 complex and should be acknowledged. As other air pollutants in the UK continue to decline
488 thanks to emission controls, it seems likely that the importance of O₃ will grow. Future work
489 on the health impacts of air pollution episodes in summer coinciding with heat should
490 consider potential worsening effects associated with local increasing O₃ concentrations, and
491 may need to be considered when planning for public health protection during these periods.

492

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511

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