Imperial College London Projects

Environmental Research Group

Pathway to WHO: achieving clean air in the UK.

Modelling air quality costs and benefits.

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1. Executive Summary

1.1. Background

The UK Environment Bill is currently under consultation, with the intention of setting two targets for $PM_{2.5}$ for 2030, one related to meeting an annual average concentration everywhere where people may be exposed, and the second relating to population exposure reduction over time. Target setting in the Environment Bill refers to the World Health Organisation (WHO) guidelines, which are themselves set by considering the scientific evidence of the human health impacts of each pollutant, but without consideration of the practicality of meeting the target. The previous and long-standing WHO guideline value was set at an annual average of 10 $\mu g\ m^{-3}$, hereafter referred to as WHO-10, but during the project has been reduced to 5 $\mu g\ m^{-3}$ (WHO Global AQ Guidelines[1]), with the original 10 $\mu g\ m^{-3}$ guideline value now considered to be an interim target. The new lower guideline value of 5 $\mu g\ m^{-3}$ reflects the increasing evidence of $PM_{2.5}$ health effects at very low concentrations.

1.2. Purpose of this report

This report was funded by the Clean Air Fund (CAF) under the project 'Pathway to WHO: achieving clean air in the UK', with the intention of submitting the results as part of the UK Environment Bill consultation.

This report provides both a technical backup to the main CAF report, and addresses the questions of whether the UK can achieve the WHO-10 target by 2030, what requirements this places on UK policy makers, and whether the costs and benefits justify such action. Within this document we specifically answer the questions:

- Can the UK meet the WHO-10 interim target in 2030?
- What population exposure reduction can be achieved between 2018 and 2030?
- What are the health benefits associated with achieving the WHO $PM_{2.5}$ interim target?
- What are the monetary benefits and costs associated with achieving the WHO $PM_{2.5}$ interim target?

To predict $PM_{2.5}$ from a base year in 2018 to 2030 we have used the NERC funded CMAQ-urban model, which couples the USEPA CMAQ and ADMS-Roads model, combined with European and UK emissions. A further London-specific modelling exercise was undertaken using Imperial's London Toolkit model, testing three scenarios, and based upon the London Environment Strategy (LES), the Major of London's roadmap for $PM_{2.5}$ and the Port of London Authority's air-quality strategy.

2030 emissions predictions included DEFRA's Business as Usual (BAU) forecast, a 'conservative' estimate of future emissions changes from UK sources, combined with widespread electrification of the UK vehicle fleet, taken from the Climate Change

[1]https://apps.who.int/iris/handle/10665/345329 (Accessed 09 February 2022).

Committee's UK 6th Carbon Budget forecast and including London's Scenario 1 (LS1), and is hereafter referred to as UK2030+LS1.

Scenario UK2030+LS1 included road traffic flow and vehicles fleet changes, as well as reductions in emissions from cooking, wood burning, construction machinery, domestic and commercial heating, railways/ships and aviation, agriculture and small-scale waste burning. Two additional London specific scenarios were tested. Scenario 2 (LS2) added further reductions to cooking and domestic wood burning, a ban on burning oil and coal, and reductions in small scale waste burning. Scenario 3 (LS3) assumed 100% reduction of domestic wood burning.

1.3. Can the UK meet the WHO-10 interim target in 2030?

UK $PM_{2.5}$ concentrations for UK2030+LS1 were forecast to be below the WHO-10 for a large proportion of the UK population. However, results showed that there were exceedences in London, close to roads and towards the city centre, as well as exceedences in other UK cities, again close to major roads.

Near to sites of industrial biomass burning, exceedences of WHO-10 occurred in 2030, although model sensitivity tests showed that this was likely to be a worst case prediction. These areas of high $PM_{2.5}$ were very local to the industrial sources and are often in locations away from large populations.

The London scenario LS2 was shown to be effective at further reducing $PM_{2.5}$ below WHO-10, with <1% of the area of London predicted to be above 10 $\mu g\ m^{-3}$. Scenario LS3 proved to have modest benefits over scenario LS2. A detailed analysis of the kerbside concentrations along London's major roads, showed that ~11% still risked having concentrations > 10 $\mu g\ m^{-3}$ for scenarios LS2 and LS3.

It is important to consider model uncertainty in interpreting the 2030 predictions. To do this we estimated a concentration below which we were 95% confident that the 2030 concentration would be below WHO-10. The concentrations were 7.9 $\mu g\ m^{-3}$ in the UK and 8.3 $\mu g\ m^{-3}$ in London.

Considering model uncertainty resulted in ~4% of the UK remaining at risk of exceeding WHO-10 (UK2030+LS1). Whilst this was a small percentage of the UK's total area, it represented the large urban populations in the south east of England and cities such as Birmingham and Manchester. For scenario LS1 the proportion of London's area at risk of exceeding the WHO-10 was 27% but for scenarios LS2 and LS3 was similar to the UK at ~4%.

Finally, recent measurements have shown that the impact of the COVID lockdown has resulted in compliance with WHO-10 in 2020 at all but a small number of sites in London.

1.4. What population exposure reduction can be achieved between 2018 and 2030?

Population Weighted Average Concentrations (PWAC), were calculated for all of the UK's 382 local authorities. PWAC links air pollution concentrations with population data, so is more relevant to the air pollution to which people are exposed.

Between 2018 and 2030, PWACs for $PM_{2.5}$ were predicted to reduce by a range of -0.9 $\mu g\ m^{-3}$ (Scotland) to almost -4 $\mu g\ m^{-3}$ (inner London) and by ~-2 $\mu g\ m^{-3}$ for the UK. By region, between 2018 and 2030, the PWACs reduced by -2.3 $\mu g\ m^{-3}$ (-23%), -1.4 $\mu g\ m^{-3}$ (-20%), -0.9 $\mu g\ m^{-3}$ (-17%) and -1.7 $\mu g\ m^{-3}$ (-23%) in England, Northern Ireland, Scotland and Wales, respectively. The PWAC in London (Scenario LS2) was predicted to reduce more, by -3.3 $\mu g\ m^{-3}$ (-29%) overall; by -3.7 $\mu g\ m^{-3}$ (-31%) in Inner London and -3.0 $\mu g\ m^{-3}$ (-28%) in Outer London. Reductions in Greater Manchester and Glasgow City were -2.6 $\mu g\ m^{-3}$ (-24%) and -1.5 $\mu g\ m^{-3}$ (-20%), respectively.

When weighted by the number of people at risk, 41% of local authorities had $PM_{2.5}$ exposure levels above WHO-10 in 2018. This was predicted to fall to less than 1% by 2030 for scenario UK2030+LS1. Furthermore, the 2030 LS2 and LS3 forecasts show that all local authorities' PWACs were under WHO-10 in 2030.

1.5. What are the health benefits associated with achieving the WHO $PM_{2.5}$ interim target?

The UK2030+LS1 scenario leads to 11.5 million life years gained across the UK population over the time period 2018–2134[2] compared with 2018 concentrations remaining unchanged. This calculation is for deaths from all causes including respiratory, lung cancer and cardiovascular deaths.

The result can also be expressed as an average gain in life expectancy of 8–9 weeks for the 745,000 children born in 2018, although this only reflects a small proportion (115,000 life years) of the overall gains in life years for children born in all the other years and for all the other age groups in 2018. As this gain in life expectancy is an average, life-expectancy gains could potentially be larger across fewer people, with the remainder less affected.

Many of the life years gained in the UK2030+LS1 scenario are in cities, including 2 million life years in London, 630,000 in Manchester and 90,000 in Glasgow from 2018-2134.

The remaining policy scenarios only involve benefits in London as that is where the additional policies are concentrated. There is predicted to be a gain of around 2.4 million life years for UK 2030+LS2 and around 2.5 million life years for UK 2030+LS3 compared with the 2 million life years for UK 2030+LS1 from 2018-2134. These figures are equivalent to a 28.5%, 29.3% and 24% reduction in life years lost respectively compared with 2018 concentrations remaining unchanged. Put another way, the additional policies in London in

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[2] The assessment of changes in life years has to be done over a long time-period because life years cannot be calculated until the population deaths have occurred.

LS2 and LS3 add 0.4 and 0.5 million life years, respectively, to the life years gained under LS1.

The improvement in average life expectancy from birth in 2018 in London is around 2–2.5 months[3] under UK2030+LS1, and 2.5–3 months for UK 2030+LS2 and LS3[4].

The gain in life years is the dominant part of the health benefits but other health outcomes were also calculated in a more approximate way. These analyses also showed substantial health benefits from both the $PM_{2.5}$ reductions and from reductions in PM_{10} and NO_2 that occurred as a consequence of the policies that reduced concentrations of $PM_{2.5}$.

The benefits from reductions in other health outcomes for the UK2030+LS1 scenario ranked by average numbers of cases per year from 2018-2030 was as follows:

- 388,000 fewer asthma symptom days in children
- 149,000 fewer adults with chronic phlegm
- 98,000 life years gained
- 25,000 fewer asthmatic children with bronchitic symptoms
- 13,000 fewer acute bronchitis infections in children
- 3,600 fewer respiratory hospital admissions
- 3,100 fewer new cases of coronary heart disease
- 2,700 fewer cardiovascular hospital admissions
- ~20 fewer infant deaths

Of course, these health outcomes vary in severity with new cases of coronary heart disease and respiratory/cardiovascular hospital admissions being more serious than symptoms. In addition, the evidence and quantification methods are more established for outcomes such as hospital admissions than for infant deaths.

The equivalent numbers for the UK2030+LS2 scenario are given in the main report. As for the gains in life years, there are improved absolute benefits for these two scenarios. The proportionate increase is relatively small (e.g. 2% fewer asthmatic symptom days in asthmatic children) because for the UK2030+LS2 scenario there are no additional UK policies and the LS1 policies have already contributed a substantial amount to air pollution reductions in London.

The additional benefits for the UK2030+LS3 scenario are only minor as a substantial reduction in wood burning has already occurred in the other scenarios and it is only one pollutant and policy addressed in this last scenario. As only $PM_{2.5}$ concentrations change, the pattern of health outcomes contributing to the benefits is a bit different. For example, with no changes in NO_2 concentrations there are no additional reductions in bronchitic symptoms in asthmatic children.

[3] The life expectancy gains in London are greater than for the UK because a proportionately greater gain in life years compared with the UK average is divided by a smaller number of births in London vs the UK.

[4] The pollution change from LS2 to LS3 is fairly small so the difference does not show at the level of rounding given here.

1.6. What are the monetary benefits and costs associated with achieving the WHO $PM_{2.5}$ interim target?

This study uses economic valuation tools to estimate the impact of air pollution across four channels: premature mortality, morbidity, healthcare costs and impacts on the labour market. The total health and economic benefits of reducing air pollution in the UK are valued at £383 billion between 2018-2134, which justifies policies that cost up to this level. Avoiding premature mortality provides the largest benefit, valued at £218 billion, while reducing the level of illness in the population across the range of disease modelled provides benefits of £130 billion. Air pollution related illnesses can result in people taking time off work (absenteeism), or attending work but being less productive (presenteeism). Reducing air pollution related illnesses could add £27 billion by reducing workplace absences and improving productivity.

This study did not assess the costs of new policies analysed. A review of the cost-benefit analysis of key policies in the UK2030 scenario including the Industrial Emissions Directive, the Medium Combustion Plant Directive, transport policies, and regulations covering wood burning and coal show that the benefits are more than two times the costs of the policies. The benefit-cost ratios for policies covering the buildings sector are typically lower, but the benefits still outweigh the costs.

1.7. Policy impacts and recommendations

$PM_{2.5}$ results for scenario UK2030+LS1 demonstrated important air pollution benefits that will improve people's health in the UK, reduce climate impacts and help achieve Net Zero commitments.

Source apportionment results for London have demonstrated that local emissions may contribute to $PM_{2.5}$ by up to ~4 $\mu g\ m^{-3}$, and that cooking, domestic wood burning and road traffic emissions were important. Finally, industrial biomass burning was an important albeit highly uncertain source of $PM_{2.5}$ UK wide.

Overall, the results for London demonstrate the benefits that may be achieved by local action and supports DEFRA's plan to combine the benefits of UK emissions reductions with local authority action to reduce $PM_{2.5}$ exposure in their area.

For robust policy assessment there is a need to reduce model uncertainty, and that PM emissions sources such as vehicle non-exhaust, domestic wood burning and cooking are likely to be important in achieving this.

A more comprehensive assessment of wood burning in industry is needed as none of the industrial biomass burning sources identified in the National Atmospheric Emissions Inventory was supported by local $PM_{2.5}$ measurements.

1.8. Discussion

This report has addressed the impacts of future emissions changes which stem from existing legislation and net zero forecasts. The forecasts we have tested achieve considerable benefits in terms of people's health. However, a considerable change in the vehicles that people drive and associated infrastructure is needed to achieve some of these benefits.

Our 2018 model evaluation resulted in good agreement with ground based $PM_{2.5}$ measurements, albeit with a small positive bias. Prediction of PM components from UK Acid Gas and Aerosol Network (AGANET) measurements also demonstrated good agreement with nitrate, sulphate and ammonium aerosols. Our results were consistent with work undertaken previously for DEFRA, based upon the year 2012 which showed similar good agreement with measurements.

However, there remain important uncertainties in both the emissions and air pollution modelling forecasts, as well as in the costs and health benefits analysis. A comprehensive uncertainty assessment is beyond the scope of this project, however we have undertaken a calculation of the areas 'at risk' of exceeding WHO-10 to help in interpreting our 2030 predictions. This leads to small areas of the UK at risk of exceeding WHO-10, although they represent highly populated areas.

Further uncertainties, in addition to the range of uncertain emissions sources listed above, remain, including uncertainty surrounding any future forecast of emissions change, potential weaknesses in the air pollution model's ability to address non-linear PM chemistry such as for secondary organic aerosol, and the impact of inter annual variation, that is, the effect of starting at a different base year, such as the exceptional 2003. Additionally, it is important to note that this is just one of a number of possible future forecasts which we hope can contribute to the evidence base for the UK Government's consultation exercise.

Whilst it is challenging to predict compliance with a specific target, in this report we have used the Natural Environment Research Council (NERC) funded CMAQ-urban, model, coupling the sophisticated United Stated Environmental Protection Agency's (USEPA) Community Multiscale Air Quality (CMAQ) model with the Atmospheric Dispersion Modelling System (ADMS) local-scale roads model. This model has been linked to well established emissions from the UK National Atmospheric Emissions Inventory (NAEI), a UK road emissions model using London Atmospheric Emissions Inventory (LAEI) methods, and published estimates of European emissions and global boundary conditions. We have provided details regarding our model set-up, and have been subject to an independent evaluation of the model via DEFRA's model inter-comparison exercise (MIE), which is due to publish shortly. We have demonstrated the impact of model uncertainty in meeting targets, and have compared our results with other DEFRA model forecasts in 2030. Where we have found high concentrations of $PM_{2.5}$ from industrial biomass burning we have tested our model assumptions to further understand the results that we obtained, concluding that they are a worst case.

The results in this report show substantial health benefits, particularly for the UK2030+LS1 scenario with increased benefits for the UK2030+LS2 scenario with benefits marginally larger again for UK2030+LS3.

One obvious question is whether the estimates could possibly be accurate when predicting health so far into the future. However, we know for sure that the benefits will be underestimated if cut short at, for example, 2030. The air pollution reductions could have contributed to less initiation of disease and avoidance of mortality that would have occurred beyond 2030. Birth and mortality rate projections have been incorporated to cover one aspect of future trends. It is also likely that further policies for further reductions will be developed beyond 2030, at which point the analyses will be repeated. So, the process is best seen as predicting into the future to the best of our ability with constant updates over time.

Some of the health outcomes quantified for this report have a long history of quantification. Others are well established health outcomes, but less commonly quantified. This is partly because assumptions have to be made about baseline rates such as symptoms days which are not routinely collected. Other areas of evidence have become established in recent years (e.g. incidence of coronary heart disease) but quantification methods are not fully developed. Further thinking is needed as to how to deal with diseases that are risk factors for each other such as coronary heart disease and stroke. And other evidence such as that on dementia may become further established to allow inclusion in the future.

None of the above uncertainties take away from the fact that air pollution reductions aimed at attaining the 2005 WHO guideline for $PM_{2.5}$ are likely to deliver substantial health benefits.

1.9. Acknowledgments

We would like to thank the contributions made by Yvonne Brown and Erwan Corfa at Transport for London (TfL), Rosalind O'Driscoll at the Greater London Authority (GLA) and Veronica HG Chan at the Port of London Authority (PLA) in developing the London emissions scenarios. We would also like to acknowledge the contribution of Eoin Devane and David Joffe at the Climate Change Committee (CCC) for their help with the vehicle forecasts, Vivid Economics for their analysis of the health economic impacts and Shawn Lee for help with references.

2. Introduction

2.1. Background

The UK Government's Environment Bill is currently under consultation and requires that a minimum of two legally binding air quality targets are set; an annual mean $PM_{2.5}$ concentration target and at least one long-term (greater than 15 years) target. Defra have proposed that the long-term air quality target is a population exposure reduction target, and that the two $PM_{2.5}$ targets would work together to provide equity (by bringing down hotspots) and continuous improvement in public health (by driving action where it is most beneficial). The new targets apply at a national level, with local authorities having a role in delivery which will be reviewed every five years. Non-legally binding interim targets also need to be set.

The results presented here supports a Clean Air Fund report, which will be submitted as part of the Environment Bill consultation on target setting, and addresses the questions:

- Can the UK meet the WHO-10 interim target in 2030?
- What population exposure reduction can be achieved between 2018 and 2030?
- What are the health benefits associated with achieving the WHO $PM_{2.5}$ interim target?
- What are the monetary benefits and costs associated with achieving the WHO $PM_{2.5}$ interim target?

Described in this document are two modelling exercises, a UK model assessment to establish an overall change in $PM_{2.5}$ between 2018 and 2030, and a separate model assessment of policy options in London, the city which may have the most difficulty in achieving the WHO interim target. Experience in London may be useful for other cities considering a similar approach to meeting the WHO target.

2.2. UK and London scenarios

UK Scenario

For the UK air quality forecast between 2018 and 2030 the emissions used were a combination of DEFRA's BAU scenario and the Climate Change Committee's (CCC) Balanced Net Zero Pathway (BNZP) for vehicle emissions, which includes a rapid transition to electric vehicles. DEFRA's BAU forecast is based upon existing environmental policies (e.g. the Industrial Emissions Directive, Euro standards for vehicles), summarised in the 2018 Energy and Emission Projections[5] (EEP). For some of the sources, 2030 emissions were adjusted to account for methodological changes in the NAEI.

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[5] https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/794590/updated-energy-and-emissions-projections-2018.pdf (Accessed 09 February 2022).

London scenarios

After undertaking a model evaluation in 2018, the 2030 forecasts in London were split into three scenarios to help establish how local policies may reduce $PM_{2.5}$ below the WHO-10. The scenarios were in addition to the UK emissions reductions and include:

- Scenario LS1, which is based upon the London Environment Strategy[6] (LES) and includes road traffic changes, such as smaller vehicle km estimates compared with the UK assumptions, the phasing out of diesel buses and taxis and small changes to vehicle electrification compared with the UK CCC BNZP. Scenario LS1 also has reductions in emissions from cooking, wood burning, Non-Road Mobile Machinery (NRMM), domestic and commercial gas/coal and oil combustion, railways, aviation, agriculture, small-scale waste burning and ships, the latter using assumptions based upon the Port of London Authority's Emission Reduction Roadmap[7] and Air Quality Strategy[8]).
- Scenario LS2 extends LS1 to include additional powers required by the Mayor, tackles some non-transport sources and is based upon the Mayor's $PM_{2.5}$ roadmap document[9]. Specifically, LS2 adds further reductions to cooking and domestic wood burning, a ban on burning oil and coal, and reductions in small scale waste burning emissions.
- Scenario LS3 extends LS2 further by assuming 100% reduction to domestic wood burning.

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[6] https://www.london.gov.uk/sites/default/files/london_environment_strategy_0.pdf (Accessed 09 February 2022).

[7] https://server1.pla.co.uk/assets/emissionsroadmapjune2020final.pdf (Accessed 09 February 2022).

[8] https://server1.pla.co.uk/assets/airquality2020v1.pdf (Accessed 09 February 2022).

[9] https://www.london.gov.uk/sites/default/files/pm2.5_in_london_october19.pdf (Accessed 09 February 2022).

3. European, UK and London emissions in 2018 and 2030

3.1. European emissions in 2018 and 2030

It is important to include the impacts of long-range transport of precursor emissions to estimate pollutants such as $PM_{2.5}$. This necessarily includes emissions from Europe. European emissions of NO_X, CO, PM_{10}, $PM_{2.5}$, SO_2, HCl, VOCs and NH_3 were acquired from the European Monitoring and Evaluation Programme (EMEP) Centre on Emission Inventories and Projections (CEIP[10]) in 2018 and summarised as a set of 50km grids. Future European emissions projections for each nation state was taken from the EU's Second Clean Air Outlook[11], which was published in Jan 2021 and provides total emissions for all pollutants, by snap sector, for each nation from now until 2050. The EU emissions changes between 2018 and 2030 for each country are included in Table 1 below.

Table 1 European emissions changes between 2018 and 2030 by country

Country	NH_3	SO_2	NO_x	$PM_{2.5}$	VOC
Albania	24.5	6.3	-17.5	-23.9	-27.5
Armenia	17.7	62.7	9.2	-13.5	-26.4
Austria	-11.2	-25.2	-57.4	-39.2	-23.8
Azerbaijan	14.2	11.3	45.7	38.1	30.5
Belarus	10.7	8.0	-9.4	10.6	-16.3
Belgium	-13.8	-38.1	-45.9	-33.0	-12.6
Bosnia-H	18.4	-74.3	-38.7	-20.2	-20.2
Bulgaria	-6.6	-50.8	-36.0	-66.4	-40.1
Croatia	-18.9	-44.5	-47.7	-64.0	-26.8
Cyprus	3.0	-75.4	-46.9	-35.1	-21.4
Czech Rep.	-20.2	-45.5	-33.9	-56.2	-27.4
Denmark	-2.4	-29.0	-42.6	-57.4	-25.4
Estonia	-6.4	-31.7	-31.2	-54.8	-21.8
Finland	1.2	-35.2	-32.6	-29.9	-22.2
France	-16.0	-39.9	-52.4	-48.5	-21.4
Georgia	13.6	42.1	27.5	1.3	10.3

[10] https://www.ceip.at/ (Accessed 09 February 2022).

[11] https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52021DC0003&from=EN (Accessed 09 February 2022).

Germany	-27.0	-39.5	-48.2	-19.8	-10.5
Greece	-1.4	-57.0	-51.6	-45.2	-34.0
Hungary	-25.6	-62.7	-43.1	-65.7	-34.4
Iceland	-1.1	11.5	13.4	12.6	4.4
Ireland	6.6	-50.8	-37.5	-36.7	-10.0
Italy	-9.1	-25.7	-48.4	-54.2	-20.3
Latvia	-6.7	-11.5	-29.2	-58.0	-21.9
Lithuania	4.2	-29.8	-41.3	-68.6	-33.5
Luxembourg	-0.6	-21.7	-62.7	-12.0	-13.1
Malta	-4.5	-61.2	-48.0	-32.2	-17.3
Moldova	3.4	-6.6	-24.1	-17.3	-35.2
Montenegro	-10.9	-92.2	-44.9	-25.7	-22.1
Netherlands	-6.6	-16.8	-41.8	-14.8	-7.5
North Macedonia	-4.2	-52.5	-27.9	-8.8	-9.1
Norway	7.5	13.7	-27.3	-31.9	-6.6
Poland	-3.4	-58.7	-38.8	-53.5	-30.5
Portugal	-0.1	-34.1	-41.4	-39.7	-21.5
Romania	-5.2	-52.5	-37.1	-65.8	-43.9
Russia	6.3	-0.2	-11.5	-2.8	-7.1
Serbia	-19.8	-66.6	-32.0	-15.3	-18.3
Slovakia	-10.9	-65.5	-35.1	-53.5	-23
Slovenia	-4.8	-38.4	-45.8	-57.1	-24.4
Spain	-16.4	-60.4	-48.7	-55.8	-13.5
Sweden	-3.7	-14.6	-56.6	-24.3	-11.5
Switzerland	-0.9	8.9	-26.8	0.5	-4.0
Turkey	20.4	-26.5	-15.4	0.2	-7.7
Ukraine	4.5	-34.4	4.6	18.2	-14.4

3.2. UK emissions in 2018 and 2030

Anthropogenic emissions for the UK, including NO_X, CO, PM_{10}, $PM_{2.5}$, SO_2, HCl, VOCs and NH_3 were taken from the NAEI (v2018) and combined with Imperial's road emissions model. The emissions sources included 11 UNECE Selected Nomenclature for Air Pollution (SNAP) sources. Emissions for the snap sector, 'Other sources and sinks (nature)', was calculated separately using the CMAQ model and included soil NO_X. The spatial scale of the emissions has been tailored to work with the CMAQ-urban air pollution model, including 10 kms and 2 kms across the UK and down to emissions for individual major roads. These anthropogenic emissions were further processed into hourly gridded chemical species using scaling factors developed in the US-EU project, Air Quality Modelling Evaluation International Initiative AQMEII[12], for use with the CMAQ-urban model.

Biogenic emissions were estimated using the Model of Emissions of Gases and Aerosols from Nature (MEGAN v2.1; Guenther et al., 2012). MEGAN is a modelling framework for estimating fluxes of biogenic compounds between terrestrial ecosystems and the atmosphere using simple mechanistic algorithms to account for the major known processes controlling biogenic emissions. The minimum parameters required by the model, are plant functional type (PFT) and leaf area index (LAI). PFT was obtained from the MCD12Q1.051 MODIS/Terra and Aqua Land Cover Type (Friedl et al 2010), and LAI was obtained from MCD15A2H MODIS/Terra+Aqua leaf area index (Myneni and Park 2015).

The emissions forecasts between 2018 and 2030 were taken to be a combination of DEFRA's business as usual (BAU) scenario, a 'conservative' estimate of emissions changes (see Table 2). DEFRA's forecast was based upon existing environmental policies (e.g. the Industrial Emissions Directive, Euro standards for vehicles), and energy forecasts from the Department for Business, Energy and Industrial Strategy (BEIS) 2018 Energy and Emission Projections[13] (EEP). Note that the BAU does not reflect measures under development for the UK's Clean Air Strategy. Where the 2030 forecast for this project differs from the DEFRA BAU, is through using of the Climate Change Committee's (CCC) Balanced Net Zero Pathway for vehicle emissions, which includes a rapid transition to electric vehicles.

[12]http://aqmeii-eu.wikidot.com/ (Accessed 09 February 2022).

[13]https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/794590/updated-energy-and-emissions-projections-2018.pdf (Accessed 09 February 2022).

Table 2 UK emissions changes between 2018 and 2030 by snap sector ‑ DEFRA Business as usual

Ktonnes per annum	2018						2030					
SNAP	NH_3	SO_2	NOx	PM_{10}	$PM_{2.5}$	VOC	NH_3	SO_2	NOx	PM_{10}	$PM_{2.5}$	VOC
1	0.3	57.9	150.5	4.4	3.5	4.4	0.3	29.0	100.7	3.5	2.8	3.5
2	2.5	33.1	46.1	48.1	47.0	47.8	2.9	11.3	37.9	29.3	28.6	24.7
3	0.4	40.9	133.7	19.7	18.5	6.2	0.4	20.3	118.0	14.7	13.8	5.8
4	2.6	8.8	10.8	49.3	7.5	144.1	2.5	9.3	7.7	41.0	6.6	145.5
5	0.0	0.6	2.0	1.0	0.5	124.8	0.0	0.2	0.8	0.4	0.2	70.4
6	1.2	0.0	0.0	1.9	1.2	284.8	1.3	0.0	0.0	1.9	1.1	293.8
7*	4.4	1.3	270.1	45.6	15.5	26.1	4.9	1.3	43.1	38.9	11.0	27.7
7 London			20.1	3.0	1.1				2.3	2.5	0.7	
8	0.0	2.7	82.6	6.2	6.2	25.6	0.0	2.7	62.8	3.4	3.3	21.7
9	22.3	0.6	1.3	1.9	1.7	7.9	22.5	0.6	1.3	1.9	1.7	7.3
10	231.7	0.0	26.9	16.0	2.8	110.9	229.7	0.0	27.1	15.6	2.8	107.5
11	9.0	0.0	0.2	3.9	3.3	0.7	9.4	0.0	0.2	3.8	3.1	0.7
Domestic Ships	0.0	11.1	74.6	1.7	1.6	18.8	0.0	5.9	57.4	1.3	1.2	18.8
International Ships	0.0	115.4	665.2	14.5	13.8	0.0	0.0	60.7	601.7	11.4	10.8	0.0

*Note snap 7 emissions changes for NO_X, NO_2, PM_{10} and $PM_{2.5}$ are from Imperial's UK vehicle emissions model

Snap sector description: Snap 1 - Combustion in the production and transformation of energy, **Snap 2** - Non-industrial combustion plants, **Snap 3** - Industrial combustion plants, **Snap 4** - Industrial processes without combustion, **Snap 5** - Extraction and distribution of fossil fuels and geothermal energy, **Snap 6** - Use of solvents and other products, **Snap 7** - Road Transport, **Snap 8** - Other mobile sources and machinery, **Snap 9** - Waste treatment and disposal, **Snap 10** – Agriculture, **Snap 11** – Other sources and sinks (nature) is calculated using the CMAQ model.

The BEIS EEP estimates include a large number of policies focused on reducing climate impacts, such as building and vehicle energy efficiency. From it we have identified a subset of EEP policies that are most relevant for air pollution and these are summarised in Table 3.

Table 3 Subset of EEP policies that also have an impact on air pollution

Sector	Policy
Energy and Industry	**Industrial Emissions Directive** (replaced the Large Combustion Plant Directive) **Medium Combustion Plant Directive** covering air pollution from mid-sized combustion plants not captured by the IED (between 1 and 50 MW)
Transport	**EU exhaust emission standards** which regulate emissions from new vehicles sold in the UK including Euro 6 implemented in 2015. Government policies to **decarbonise transport** and achieve net-zero are not yet fully fleshed out. We have therefore drawn upon CCC estimates from the Sixth Carbon Budget
Buildings	**Building regulations** updated in 2010 and 2013 which sets minimum energy performance standards. **Products policy** (EU Ecodesign Directive and Energy Labelling Framework Regulation) sets minimum performance and information requirements for energy-using products. **Renewable Heat Incentive** provides financial incentives to increase the uptake of renewable heat for non-domestic and domestic users. **Heat Network Investment Project** provides capital funding in England and Wales to encourage the development of heat networks. **Private rented sector Energy Efficiency Regulations** require privately rented properties to have a minimum energy performance rating of E. **Boiler Plus** aims to deliver additional energy and carbon savings from the domestic heating sector in England by lowering overall gas demand from domestic properties. **Regulations covering the sale of wet wood and traditional coal** that regulates the sale, distribution and marketing of bituminous coal and wet wood, and places limits on the sulphur content of smokeless fuels with the aim of improving air quality.
Agriculture	Conversations with Defra and Ricardo suggest that agriculture policies are not included in the baseline.

There are other EEP policies that are likely to have a positive impact on reducing air pollution. For example, policies that encourage fuel switching (e.g. to renewables) and investment in energy saving technologies. There is also a combination of UK and EU policies that strengthen the business environment to invest in such measures (e.g. EU ETS, Contracts for Difference, carbon price floor). In addition, there is the Renewable Transport Fuel Obligation, which has been instrumental in increased biofuel use.

3.3. Defra 2030 BAU emissions adjustments

The DEFRA BAU emissions were updated prior to the UK model runs being undertaken, with the updates reflecting a number of recent methodological changes in the NAEI and affecting the 2030 forecasts. A detailed list of those changes is given in Table 4, below.

Table 4 A list of assumptions included in the DEFRA BAU, by snap sector, plus the adjustments made to sources where applicable

Snap	NFR - Sector	Adjustments to BAU
1,2	1A1a - Power Stations	Power stations – Adjustment for new natural gas projection data - The original emission projections developed for NAEI were adjusted to reflect the more recent generation projections from BEIS (2020). An adjustment was also included to reflect the differences in gas generation between EEP2019 and the latest generation estimates from BEIS. The baseline projections have been adjusted to also account for the BAT conclusions for Waste Incinerations (WI) which will be legally binding in 2025 and 2030.
1	1A1b/c - Other Energy Ind	
2,3,8	1A2/4a/4c - Other Stationary comb	MCPD & HNG Regulations - The NAEI does not fully account for the impact of the Medium Combustion Plant Directive (MCPD) and the High NO_X Generators (HNG)
7	1A3b - Road Transport	Euro 6 diesel adjustment - Based on data from COPERT v5.4, the emission factor for NOx from certain Euro 6 diesel cars in real world conditions is expected to be lower compared to the emission factors previously provided in COPERT v5.3 (which the NAEI projections are based upon). The total NOx emissions from Euro 6 diesel cars is likely to be an overestimate, so a baseline adjustment has been applied.
8	1A2gvii, 1A3eii, 1A4bii, 1A4cii - NRMM	Directive 97/68/EC on emissions from non-road mobile machinery (NRMM) - To account for the revision of Directive 97/68/EC on emissions from NRMM engines (proposed measure Introducing Stage control limits for <18kW industrial off-road machinery, setting a limit of 7.5 g/kW for NOx and 0.4 g/kW for PM. Phase out of the use of Red Diesel - The NAEI does not fully account for the removal of the entitlement to the use of gas oil, otherwise known as red diesel, from April 2022 for all users except for agriculture, rail and for non-commercial heating (Finance Bill 2021).
8	1A3dii - Ships	Phase out of the use of Red Diesel - The NAEI does not fully account for the removal of the entitlement to the use of gas oil, otherwise known as red diesel, from April 2022 for all users except for agriculture, rail and for non-commercial heating (Finance Bill 2021).
2,4	1A4bi - Domestic comb	Defra new domestic wood burning activity and emission factors for wet wood. Legislation regulating the sale of wet wood and traditional coal in England - The sales of house coal and wet wood in England was phased out in May 2021, with transition periods available.

3.4. Non-road transport emissions in London

We have based our London base case 2018 emissions and the three 2030 scenarios on the London Atmospheric Emissions Inventory (LAEI2016[14] and LAEI2019[15]), which includes detail of estimates of all anthropogenic emissions sources. Whilst the LAEI has similar outputs to the NAEI, such as 1x1km estimates of annual emissions of NO_X, NO_2, $PM_{2.5}$, PM_{10}, SO_2, CO and VOCs, it has some notable differences. Of particular importance are the road emissions, described below, which are calculated 'bottom up', that is road by road, and includes specific London vehicle stock, not available from the NAEI. The LAEI also has a specific domestic wood burning PM emissions inventory, which is potentially more realistic, since it is reflected in an analysis of measurements of these sources in and around London; as well as an estimate of cooking emissions, which is also reflected in the measurement of cooking organic aerosols. To date cooking is not available in the NAEI. Both domestic wood burning and cooking PM emissions were calculated using multiple runs of the London air pollution model, whilst adjusting London emissions to obtain the closest agreement with measurements. The total emissions were redistributed using surrogate dwelling stock categorised by the property build period[16] and types[17] for domestic wood burning and, for cooking, using a combination of food industry sector employment[18] and commercial catering premises outlets from OpenStreetMap in the capital.

3.5. UK and London road transport emissions in 2018

The NAEI does not provide road by road emissions estimates across the UK and this limits our ability to look at air pollution concentrations close to roads, where some of the highest concentrations occur. As a consequence, researchers from the Environmental Research Group (ERG) have developed a UK emissions tool to generate annual emissions for NO_X, NO_2, $PM_{2.5}$ and PM_{10}, road by road. The road emissions are represented as points every 10m across the UK road network, using the same methods as in the London Atmospheric Emissions Inventory (LAEI2019[19]). Traffic counts were broken down into 11 vehicle types and split by vehicle age/Euro Standard and fuel type and used in conjunction with COPERT v5.4 emissions factors.

Briefly, the following data sources were used to create the UK bottom-up road emissions estimates:

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[14] https://data.london.gov.uk/dataset/london-atmospheric-emissions-inventory--laei--2016 (Accessed 09 February 2022).

[15] https://data.london.gov.uk/dataset/london-atmospheric-emissions-inventory--laei--2019 (Accessed 09 February 2022).

[16] http://ubdc.gla.ac.uk/dataset/property-build-period-lsoa/resource/d022a431-1687]-422e-ae53-fca9ec221c45 (Accessed 09 February 2022).

[17] http://ubdc.gla.ac.uk/dataset/property-type-lsoa (Accessed 09 February 2022).

[18] https://data.london.gov.uk/dataset/pubs-clubs-restaurants-takeaways-borough (Accessed 09 February 2022).

[19] https://data.london.gov.uk/dataset/london-atmospheric-emissions-inventory--laei--2019 (Accessed 09 February 2022).

- DfT Major road activity data on 55,000 km of roads in the UK, expressed as Annual Average Daily Totals (AADT), and in London from TfL. This data included an emissions profile applied for specific vehicle groups, based upon their diurnal activity patterns in a given location. Traffic counts were disaggregated into vehicle groups (i.e. car, motorcycle, taxi, light goods vehicle (LGV), rigid heavy goods vehicle (HGV), articulated HGVs, bus and coach), with HGVs further split by the number of axles;
- Major road speed data, for each road and for some at sub road link level was used to calculate the road by road emissions.
- Minor road activity data – annual vehicle km estimates, that are not specifically accounted for on the major road network, were assigned to the minor roads by km^2 and vehicle group, with speed assumptions based upon location to enable emissions from these sources to be calculated.
- Cold start emissions were calculated according to the methods set out in the EEA guidebook[20], using the number of vehicle starts in each km^2 according to data provided by Transport for London (TfL) in London, or otherwise in the rest of GB using the UK's National Trip End Model. Cold start emissions were added, as tonnes/annum onto the km^2 grid totals.
- Hot exhaust emissions factors were taken from COPERT v5.4
- Non-exhaust emissions factors of PM_{10} and $PM_{2.5}$ were taken from a combination of the EMEP emissions guidebook, rescaled using data from Harrison et al. (2012). In it there are separate factors for Tyre wear, brake wear and resuspension/surface wear. A brief description of the method is given in the LAEI 2010 documentation[21].
- Vehicle km derived fleet composition data, which was split by road type in the UK and taken from the Base 2020 (NAEI 2018) data, covering differences in fuel type (i.e. petrol, diesel, electric) and Euro standard by each vehicle class.
- In London, iBus data was used to provide bus route-specific flows, fleet composition and speed.

By combining these 'bottom-up' emissions, we were able to create 1x1km annual emissions totals across Great Britain (GB) for use with CMAQ-urban. Note that Northern Ireland has a separate vehicle dataset and mapping regime and as a consequence we used the NAEI 1x1 km emissions and a more limited dataset of Northern Ireland roads, mainly in Belfast.

3.6. Exhaust and non-exhaust road transport emissions in 2030

Future changes to vehicle exhaust emissions were estimated using the methods described above, combined with the CCC's estimate of vehicle km from the BNZP (Table 5), projected Euro standards for different vehicle types from the NAEI, and COPERT v5.4 emission

[20] https://www.eea.europa.eu/publications/emep-eea-guidebook-2016/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/1-a-3-b-i (Accessed 09 February 2022).

[21] https://data.london.gov.uk/dataset/london-atmospheric-emissions-inventory-2010 (Accessed 09 February 2022).

factors. The assumptions related to the BNZP of the CCC are discussed in detail elsewhere (see CCC report[22]), but are briefly described below.

Table 5 UK Billion Vehicle km changes between 2020 and 2030 from the Climate Change Committee – Balanced Net Zero Pathway

UK Billion Vehicle kilometres		2020	2025	2030
cars	ICE/HEV - petrol	184.9	153.8	101.3
cars	ICE/HEV - diesel	241.7	200.2	128.7
cars	PHEV - petrol	1.7	15.9	18.2
cars	PHEV - diesel	2.1	20.8	23.6
cars	BEV	3.5	51.2	183.4
cars	H2FC	0.0	0.0	0.0
Van	ICE/HEV - petrol	0.0	0.0	0.0
Van	ICE/HEV - diesel	85.1	78.8	56.2
Van	PHEV - petrol	0.0	0.0	0.0
Van	PHEV - diesel	0.8	1.0	0.9
Van	BEV	0.9	11.9	41.6
Van	H2FC	0.0	0.0	0.0
Motorcycle	ICE/HEV - petrol	4.5	4.5	3.7
Motorcycle	ICE/HEV - diesel	0.0	0.0	0.0
Motorcycle	PHEV - petrol	0.0	0.0	0.0
Motorcycle	PHEV - diesel	0.0	0.0	0.0
Motorcycle	BEV	0.0	0.2	1.3
Motorcycle	H2FC	0.0	0.0	0.0
HGV	ICE/HEV - petrol	0.0	0.0	0.0
HGV	ICE/HEV - diesel	27.9	26.0	24.4
HGV	PHEV - petrol	0.0	0.0	0.0
HGV	PHEV - diesel	0.0	0.0	0.0

[22]https://www.theccc.org.uk/wp-content/uploads/2020/12/Sector-summary-Surface-transport.pdf (Accessed 09 February 2022).

HGV	BEV	0.0	0.1	0.8
HGV	H2FC	0.0	0.0	0.2
Bus	ICE/HEV - petrol	0.0	0.0	0.0
Bus	ICE/HEV - diesel	4.1	3.9	3.4
Bus	PHEV - petrol	0.0	0.0	0.0
Bus	PHEV - diesel	0.0	0.0	0.0
Bus	BEV	0.0	0.2	0.3
Bus	H2FC	0.0	0.0	0.4
Total all vehicles		**557.2**	**568.5**	**588.4**

Zero emission vehicle uptake. Battery Electric Vehicles (BEV) make up the majority of new car and van sales by 2030, while Heavy Duty Vehicle (HDV) sales, a mixture of BEVs and hydrogen vehicles, ramp up during the 2030s.

- The CCC assumed that sales of new petrol and diesel cars and vans are phased out by 2032. BEV ranges increase, while battery cost reduces from around £121/kWh today to £48/kWh by 2030 and £44/kWh by 2040. As a result, BEVs make up 48% of all new sales in 2025, 97% in 2030 and 100% from 2032 onwards.
- Plug in Hybrid Electric Vehicles (PHEV) sales increase in the short term, reaching 25% in 2025, before falling to near zero by 2030.
- Commercial-scale zero-emission HDV trials take place from the early-2020s. Infrastructure development continues for the most cost-effective solutions, assumed to be batteries and hydrogen initially. Government subsidies ensure Total Cost of Ownership (TCO) parity between zero emission and diesel options in 2035. As a result, BEVs make up 12% of new HGV sales and 25% of new bus sales in 2030, rising to 51% and 44% in 2040. Hydrogen fuel-cell vehicles make up 7% of new HGV sales and 44% of new bus sales in 2030, and 48% and 55% in 2040.

Efficiency and biofuels. New conventional vehicles become more fuel efficient. Biofuels have a role in reducing emissions from the remaining petrol and diesel vehicles during the transition to ZEVs.

- The carbon intensity of new conventional vehicles improves. HGVs have efficiency savings, ranging from 13% for small rigid trucks to 22% for large articulated vehicles. Uptake of these measures reaches 80% of HGVs from 2025.
- Following the introduction of E10 (10% ethanol) in 2021, biofuels make up around 7% (by energy) of the conventional fuel used by cars and vans.
- Among HDVs, the proportion of biofuels in the diesel consumed rises from 4% in 2030 to 12% by 2040.

Demand reduction†††. Demand for car travel is reduced by a combination of societal and technological changes reducing the need for travel and modal shift. Logistics and operational improvements reduce HGV demand.

- Average car-kilometres decrease by 6% by 2030. Demand reduction for vans is lower, reaching 3% from 2030 onwards. Improved speed limit enforcement gives efficiency savings of 2% from 2025.
- Factors including improved logistics mean that demand reductions for HGVs increase gradually to 10% for rigid HGVs and 11% for articulated HGVs by 2030, remaining at these levels thereafter

(††† - Note that these reductions are relative to a baseline in which car ownership, and hence total car-kilometres, are assumed to be increasing. Overall vehicle-kilometres are expected to grow by 5% by 2030 and by 15% by 2050.)

Using the BNZP for 2030 vehicle emissions means that BEV and PHEV vehicles become increasingly important (Table 5) and that this has an important influence on Non-Exhaust Emissions (NEE). Briefly, whilst the emissions from vehicle exhausts have declined over recent years, non-exhaust emissions from Tyre wear, brake wear and resuspension remain relatively constant. So much so that NEE of $PM_{2.5}$ and PM_{10} currently represent the majority of vehicle emissions. However, in future, with the increased use of PHEVs and BEVs there are likely to be important changes in non-exhaust PM emissions, and of particular interest are the effects on brake wear emissions through use of regenerative braking and resuspension, through increased vehicle weight of electric vehicles compared with their Internal Combustion Engine (ICE) equivalents.

Electric and hybrid vehicles, which are heavier than ICE vehicles, are predicted to emit more NEE than conventional ICE vehicles due to the additional mass of the battery pack (Timmers and Achten 2016, OECD 2020, Beddows and Harrison 2021). However, hybrid/electric powertrains also incorporate regenerative braking systems (i.e. the energy recovery mechanism that causes resistance braking via the vehicle's motor acting as a generator to convert kinetic energy into electrical energy. To date, there has been limited published research assessing the impact of regenerative braking on NEE, although as it reduces traditional friction braking, it is expected to reduce brake wear PM_{10} and $PM_{2.5}$ emissions (OECD 2020).

To calculate how PM emissions from brake wear and resuspension change, emissions have been calculated for several passenger vehicles using ICE, PHEV and BEV powertrains. Table 6 details the vehicle classes, powertrains, mass, brake technology, and recuperation potential used to determine the changes in NEE. Subcompact (SC) and large-sport utility (L-SUV) vehicles have been used, since they represent a sizeable proportion of the UK vehicle fleet. The mass of each vehicle has been derived from industry data, and are comparable to previous estimates of vehicle mass (Timmers and Achten 2016, Beddows and Harrison 2021, Liu, Chen et al. 2021).

Table 6 Assessed vehicle class, powertrain, mass, brake technology, and potential recuperation which have been used to determine the changes in NEE. There is up to 30% increase in mass of a BEV compared to ICE

Vehicle class	Powertrain	Braking	Technology	Regen braking (g)	Mass (kg)
Subcompact (SC)	*Conv 60 kW*	*No regeneration*		*NA*	*1175*
	48-volt HEV	10 / 15 kW EM	Vacuum Brake + ESP hev single	0.2	1214
	280-volt PHEV	40 / 60 kW EM	iBooster + ESP hev single	0.3	1251
	280-volt BEV	40 / 60 kW EM	iBooster + ESP hev single	0.3	1530
Large-sport utility vehicle (L-SUV)	*Conv 110 kW*	*No regeneration*		*NA*	*2325*
	48-volt HEV	10 / 15 kW EM	Vacuum Brake + ESP hev single	0.2	2364
	280-volt PHEV	40 / 60 kW EM	iBooster + ESP hev single	0.3	2484
	280-volt BEV	40 / 60 kW EM	iBooster + ESP hev single	0.3	2800

The impact of increased vehicle mass on NEE has been calculated using a regression-based approach developed by Beddows and Harrison (2021). In the paper, ‘Base’ brake and tyre wear PM_{10} and $PM_{2.5}$ EFs for urban, rural and motorway driving were obtained from the EMEP/EEA Guidebook, and combined with base resuspension (which includes road surface wear) EF’s derived from the US-EPA’s AP-42 document (USEPA 2011, Ntziachristos and Boulter 2019, Beddows and Harrison 2021). Note that the resuspension EF’s do not account for different types of driving (e.g. urban/rural/motorway).

Second, correlations between the base EFs and the vehicle types presented in Table 6 were determined using Eq. (1):

$$EF = b * W_{ref}^{1/C} \quad (1)$$

Where W_{ref} is the vehicle mass of the assessed vehicle category divided by 1000 kg, *b* (mg km^{-1} veh^{-1}) and *c* (no unit) are NE specific parameters used to fit the equation. This relationship is then applied to the base NEE to account for the changes in vehicle weights, see the results in Table 7.

Separately, the impact of regenerative braking has been predicted by simulating the regenerative braking and friction brake force needed for each vehicle to complete the Transport for London (TfL) and World Harmonized Light Vehicles Test Procedure (WLTP) drive cycles, using GT-SUITE software[23]. The TfL drive cycle was used to represent London emissions and the WLTP cycle to represent emissions throughout the UK. The GT-SUITE software is an industry standard physics-based model used, amongst other applications, to design vehicle braking systems, the results of which were available through ERG's collaboration with Robert Bosch ltd. GT-SUITE uses several variables including vehicle class/mass and regenerative braking technology. The drive cycles have different driving styles, speeds, and deceleration rates, with the TfL drive cycle incorporating more aggressive braking and stopping than the WLTP drive cycle.

Reductions in simulated brake force (which we have used as a proxy for brake emissions), for each electric/hybrid powertrain have been compared with the equivalent ICE vehicles to estimate the effect of regenerative braking, and the results have been averaged to represent cars, vans and taxis. The change in PM_{10} and $PM_{2.5}$ emissions based on the WLTP and TfL drive cycle are displayed in Table 7.

Table 7 Summary of changes in NEE for passenger vehicles based on changes in vehicle weight and regenerative braking (WLTP (UK) and TfL (urban))

Powertrain	Drive cycle	Brake wear		Tyre Wear		Resuspension	
		PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$
ICE	WLTP	NA	NA	NA	NA	NA	NA
HEV	WLTP	-80%	-80%	1%	3%	2%	3%
PHEV	WLTP	-91%	-92%	3%	5%	6%	8%
BEV	WLTP	-90%	-88%	10%	13%	23%	22%
ICE	TfL	NA	NA	NA	NA	NA	NA
HEV	TfL	-66%	-66%	1%	1%	2%	3%
PHEV	TfL	-69%	-69%	3%	3%	6%	8%
BEV	TfL	-67%	-66%	10%	11%	23%	22%

As this is a new but important approach to evaluating PM NEE changes, it is useful to compare the results with other published research. The reductions in brake wear emissions due to the electrification of the vehicle fleet are similar to Hooftman et al. (2016) who compared the service time of brake linings from ICE and EVs in an urban setting and suggested that friction braking reduces emissions by 66% (Hooftman, Oliveira et al. 2016). Other estimates of reductions caused by regenerative braking are summarised by the OECD (2020), and range from 25-95% reduction of EV brake wear, with less abrasion and lower disc brake system temperatures resulting in the emission reductions (Nopmongcol, Grant et al. 2017, OECD 2020, Beddows and Harrison 2021, Liu, Chen et al. 2021). However, it should be noted that the lower estimates in the literature (e.g. ≤ 50%) are based on conservative assumptions (not derived from modelling/measurements), and are unlikely to reflect the real-world benefits of regenerative braking.

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[23]https://www.gtisoft.com/gt-suite/ (Accessed 09 February 2022).

3.7. London's emissions scenarios in 2030

In London, LS1 is considered to be the business as usual scenario, i.e., this is the current best estimate of emissions in 2030, and is based upon commitments made in the London Environment Strategy. Two further scenarios extend the policies to reduce $PM_{2.5}$, and each is described in detail in the following text:

Road source category

Vehicle changes which apply to Scenarios LS1, LS2 and LS3. The two phases of the Ultra-low emission zone (ULEZ) in London were included in the forecast, although requiring only Euro 6 vehicles in the zone has limited impact in 2030, since virtually all vehicles are Euro 6. Other vehicle assumptions made in the 2030 predictions are given below in Table 8.

Table 8 Vehicle assumptions used in the London Scenarios 1-3

Category	Future forecast	Comment
Vehicle km	-5% by 2030	CCC UK vehicle growth +5%
Buses	By 2030 77.4% Electric 8.4% Hybrid Electric	Phase-out of diesel buses, and purchase of only hybrid and zero-emission double decker buses from 2018, with the entire fleet becoming zero-carbon by 2037 at the latest
Taxis	Fleet Zero emissions capable by 2033 with 19% diesel, 71% plug in hybrids and 10% electric remaining in 2030	No longer licensing new diesel taxis from 2018 and supporting the sector to upgrade to cleaner 'zero-emission capable' vehicles.
Cars	60%, 50% and 49% electric in Central, Inner and Outer London respectively in 2030	The equivalent figure from the CCC across the UK is 40%
LGV	32.5% electric in 2030	CCC's UK-wide estimate is 42%
Coaches	In 2030 are projected to be 26% electric (74% will still be diesel)	Bus and coach figures are more optimistic in London than the 17.3% UK electric vehicle figure forecast by the CCC.
Rigid and Articulated HGVs	In 2030 6% and 10% electric respectively, with the remainder still diesel	CCC UK figures are 3 and 5% respectively.
Motorcycles	27% electric by 2030, and 73% petrol vehicles.	CCC UK projection of 26% EMCs

Non-road source category

Details of the emissions changes by non-road source category are included in Table *9* below, with an additional commentary on the associated policies, taken from the LES, the Major of London's roadmap for $PM_{2.5}$ and the Port of London Authority's air-quality strategy documents, included thereafter.

Table 9 *Scenario assumptions for non-transport sources in London (tonnes/annum). Note that emissions from sources in italics remain constant between 2018/19 and 2030*

Source	**Pollutants**	**2018**	**2030 UK2030+LS1**	**2030 UK2030+LS2**	**2030 UK2030+LS3**
Commercial catering (cooking)	PM	548	479	137	137
Domestic wood burning (DWB)	PM	661	578	165	0
Construction Dust	*PM_{10}*	*2,288*	*2,244*	*2,244*	*2,244*
	$PM_{2.5}$	*229*	*224*	*224*	*224*
Construction NRMM	NO_X	1,846	368	368	368
	PM_{10}	135	13	13	13
	$PM_{2.5}$	135	12	12	12
Industrial NRMM	NO_X	427	133	133	133
	PM_{10}	37	11	11	11
	$PM_{2.5}$	37	10	10	10
Domestic Gas	NO_X	2,720	1,708	1,708	1,708
	PM_{10}	204	168	168	168
	$PM_{2.5}$	204	168	168	168
Commercial Gas	NO_X	5,485	3,611	3,611	3,611
	PM_{10}	58	39	39	39
	$PM_{2.5}$	58	39	39	39
Domestic other fuels (oil and coal)	NO_X	143	113	0	0
	PM_{10}	111	46	0	0
	$PM_{2.5}$	110	45	0	0
Commercial other fuels (oil and coal)	NO_X	2,905	1,484	0	0
	PM_{10}	217	144	0	0
	$PM_{2.5}$	188	145	0	0
Industrial Part A	*NO_X*	*3,187*	*3,187*	*3,187*	*3,187*
	PM_{10}	*48*	*48*	*48*	*48*
	$PM_{2.5}$	*46*	*46*	*46*	*46*
Industrial Part B	*NO_X*	*208*	*208*	*208*	*208*
	PM_{10}	*156*	*156*	*156*	*156*
	$PM_{2.5}$	*156*	*156*	*156*	*156*

Rail	NO_x PM_{10} $PM_{2.5}$	700 32 23	360 11 8	360 11 8	360 11 8
Shipping - Assumptions based on the PLA's Emission Reduction Roadmap[24], Air Quality Strategy[25] and 2016 emission inventory[26] reports	NO_x PM_{10} $PM_{2.5}$	890 27 26	527 16 16	527 16 16	527 16 16
Aviation Heathrow	NO_x PM_{10} $PM_{2.5}$	3,807 65 54	3,428 52 42	3,428 52 42	3,428 52 42
Agriculture	NO_x PM_{10} $PM_{2.5}$	212 43 17	155 35 14	155 35 14	155 35 14
Accidental Fires	*NO_x* *PM_{10}* *$PM_{2.5}$*	*18* *76* *71*	*17* *72* *67*	*17* *72* *67*	*17* *72* *67*
Waste SSW (Small Scale Waste) Burning	NO_x PM_{10} $PM_{2.5}$	8 100 93	8 100 93	4 50 46.5	4 50 46.5
Waste STW *(Sewage Treatment)*	*NO_x*	*505*	*505*	*505*	*505*
Waste *Landfill*	*PM_{10}* *$PM_{2.5}$*	*0.50* *0.08*	*0.50* *0.08*	*0.50* *0.08*	*0.50* *0.08*
Waste WTS *(Transfer Station)*	*PM_{10}* *$PM_{2.5}$*	*2.63* *0.26*	*2.63* *0.26*	*2.63* *0.26*	*2.63* *0.26*
Shipping *Canal, Small River*	*NO_x* *PM_{10}* *$PM_{2.5}$*	*20* *0.90* *0.90*	*20* *0.90* *0.90*	*20* *0.90* *0.90*	*20* *0.90* *0.90*

[24] https://server1.pla.co.uk/assets/emissionsroadmapjune2020final.pdf (Accessed 09 February 2022).

[25] https://server1.pla.co.uk/assets/airquality2020v1.pdf (Accessed 09 February 2022).

[26] https://www.pla.co.uk/assets/finalplaportwideinventoryoutputsreportv10.2publication.pdf (Accessed 09 February 2022).

Household Garden NRMM	*NO_X*	*68*	*68*	*68*	*68*
	PM_{10}	*1.7*	*1.7*	*1.7*	*1.7*
	$PM_{2.5}$	*1.7*	*1.7*	*1.7*	*1.7*

Small Scale Waste Burning and Commercial cooking emissions
Using new powers to require appropriate abatement of significant combustion related sources of $PM_{2.5}$ by strengthening local authority enforcement powers and conferring the ability to create zero emission zones where no combustion is allowed on certain, time limited occasions.

Domestic wood burning emissions
To address domestic wood-burner emissions through an improved testing regime, better information at the point of sale using appropriate technology/fuels for smoke control zones, and new powers for the Mayor to set tighter emission standards for wood burning stoves sold in London (for example, the eco-design standard). The Mayor will continue to work with Defra to improve the standards and testing for smokeless fuels.

Non-Road Mobile Machinery (NRMM) emissions
The Mayor will work with users of Non-Road Mobile Machinery (NRMM) to prevent or reduce NRMM emissions. Engines used in NRMM are subjected to reduced emissions limits overtime, by the EU, meaning that newer machines are less polluting than older ones. In the absence of direct powers to regulate this sector, the Mayor has issued guidance to create an NRMM Low Emission Zone through planning conditions with minimum emission standards, based on the European engine “stages”. The NRMM Low Emission Zone will include progressively tightening standards, with the current proposals as follows: Stage IV throughout London by 2025 and Stage V throughout London by 2030. The Mayor is also calling on government for new powers for regional and local authorities to control emissions from construction NRMM; this includes stronger enforcement powers to secure improved regulation of NRMM.

Domestic combustion emissions
The Mayor’s ‘Energy for Londoners’ programme will support the transition from old inefficient gas boilers to ultra-low NO_X gas boilers and alternatives, such as heat pumps. The Mayor will evaluate the boiler scrappage initiative scheme and the London Boiler Cashback and Better Boilers schemes. This will help inform the development of future initiatives to provide more efficient and low NO_X boiler replacements. Through the Energy for Londoners programme, the Mayor’s energy efficiency programmes will also help to remove inefficient heating systems that contribute to poor air quality. Oil and coal emissions will be set to zero.

Commercial combustion emissions
The Mayor will work with government to seek reductions in emissions from large scale generators producing power for commercial buildings in London. The Mayor will work with BEIS and Defra to seek market reforms and discourage the use of emergency generators in the STOR (Short Term Operating Reserve) and capacity markets. The Mayor will encourage Defra to apply more robust standards, and give the Mayor the powers to regulate this sector in London. The Mayor will also work with the retrofit industry and generator owners to develop and install effective retrofit solutions for existing generators as soon as possible.

Where applicable, retrofit for emergency generators could be supported by the Mayor's retrofit programmes.

Shipping sources

We have assumed a 40% NO_X and PM emissions reduction between 2016 (based on 2016 emission inventory[27]) and 2030. These assumptions are based on the Emission Reduction Roadmap[28] report that highlights the barriers to achieving these targets and identifies the technologies available and applicable to meet the environmental goals set out in the Port of London Authority's (PLA) Air Quality Strategy[29] document.

Rail

Rail emissions in 2030 have been based on the assumptions that rail traffic levels will have recovered from the COVID-19 pandemic by 2025 and have taken into consideration the planned and confirmed changes such as the full electrification of all services to and from Kings Cross (except for Grand Central services) and to and from Paddington; and the replacement of Voyager and Meridian trains serving Euston and St Pancras, respectively. Other projects such as GOBLIN have been treated as fully completed. Furthermore, potential growth between 2025 and 2030 have used a general projection of rail traffic growth that is similar to that observed in recent years.

Aviation

For each London airport (Heathrow and London City airports), emissions from aircraft, ground support equipment (GSE), landside vehicles and stationary sources have been projected to 2030 where relevant. Projections to 2030 were made on the basis that there are no new airport infrastructure developments nor any increases in capacity beyond existing caps on aircraft movements. Specifically, the projections assumed that there is no 3rd runway at Heathrow. The same methodology was used for the 2030 projection with the only differences relating to activity data projection, changes to the aircraft emissions brought about by the modernisation of the fleet and changes to ground vehicle fleet included newer vehicles, with tighter emissions standards, replacing older ones. All other smaller airports within London were assumed to be unchanged from 2018 on the basis that no further increases in activity levels are expected for these small airports out to 2030 and that they are small compared with Heathrow and London City airports.

[27] https://www.pla.co.uk/assets/finalplaportwideinventoryoutputsreportv10.2publication.pdf (Accessed 09 February 2022).

[28] https://server1.pla.co.uk/assets/emissionsroadmapjune2020final.pdf (Accessed 09 February 2022).

[29] https://server1.pla.co.uk/assets/airquality2020v1.pdf (Accessed 09 February 2022).

4. **UK meteorology and air pollution modelling methods**

4.1. Introduction

To calculate air pollution concentrations from the emissions described above, requires the use of air quality models, which for UK applications we refer to as CMAQ-urban (Beevers et al., 2012), and which is a combination of the Weather Researching and Forecasting (WRF V4.1)) meteorological model (Skamarock *et al* 2008), the United States Environmental Protection Agency's (USEPA) Community Multiscale Air Quality model (CMAQ V5.3.1)) (Byun and Schere, 2006), coupled with the Atmospheric Dispersion Modelling System (ADMS) roads model (CERC, 2017). Together CMAQ-urban produces air pollution from 2km grid scale across the UK, down to predictions every 20m, close to roads. In the London analysis we have used the CMAQ-urban model to provide the air pollution contribution from outside the city, but within the city have used as second model, the London Toolkit model (Beevers, 2013) to test the range of London specific emissions scenarios. We have used the London Toolkit model because it provides consistency with other policy developments such as the Ultra Low Emissions Zone (ULEZ) and the London Atmospheric Emissions Inventory, and is also quick to run.

4.2. The Weather Researching and Forecasting (WRF) model

The WRF meteorological model provides the drivers for both the UK and London dispersion models and emissions calculations, and is run across a model domain given in Figure 1. WRF output parameters include wind speed and direction, temperature, humidity and rainfall at both ground level and in layers up to approximately 15km above ground. The modelling domains consist of an outer domain with horizontal resolution of 50 km covering Europe, and two nested domains at 10 km and 2km resolution covering the UK. The model has 23 vertical layers extending from the surface up to 100 hPa, and with 7 layers within 1 km of the ground. Beyond the model domain, contributions from the northern hemisphere were taken from the NCEP final analysis (NCEP FNL, 2000) having a 6-hr time interval. The physical options used in the WRF setup were the Dudhia shortwave radiation (Dudhia, 1989), Rapid Radiative Transfer Model (RRTM) long wave radiation (Mlawer et al 1997), Kain–Fritsch cumulus parameterization scheme (Kain, 2004), Pleim-Xiu surface layer scheme (Pleim and Xiu, 2003), Rapid Update Cycle (RUC) land surface model (Benjamin, et al. 2004), and Asymmetric Convective Model version 2 (ACM2) for the planetary boundary layer parameterization (Pleim, 2007).

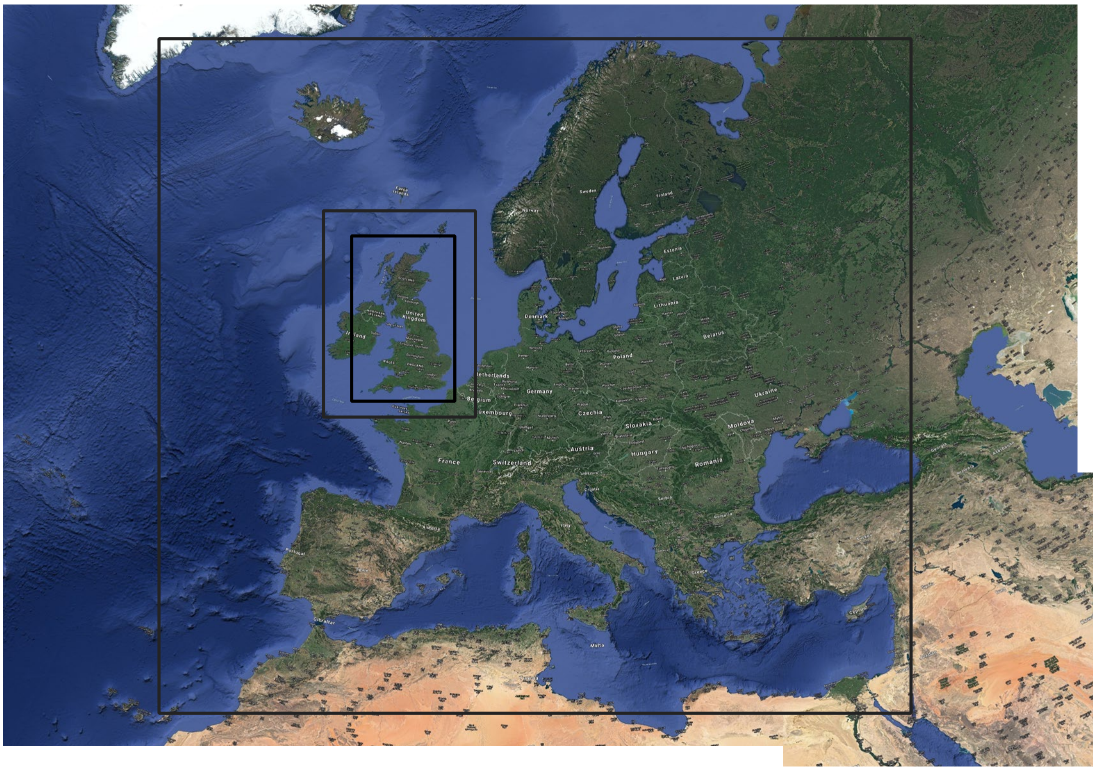


Figure 1 WRF and CMAQ model domains

4.3. The Community Multi-Scale Air Quality (CMAQ) model

The CMAQ model operates over the same model domain as WRF, representing the long-range transport of pollutants from all sources 100-1000s km away, important for pollutants like O_3 and PM. The model includes state of the science atmospheric chemistry and physics and outputs NO_X, NO_2, PM_{10}, $PM_{2.5}$ and O_3 concentrations. The carbon bond mechanism version 6 (CB6) (Yarwood et al 2010) has been used for gaseous species and predictions of aerosols employ the 7th generation aerosol module (AERO7) mechanism. The simulations were carried out for each month of 2018, and a spin-up period of 3 days from the previous month was used and then discarded.

4.4. Representing road sources within the CMAQ-urban model

The CMAQ-urban model estimates $PM_{2.5}$, PM_{10}, NO_X and NO_2 concentrations from local road traffic sources, to give the spatially detailed forecasts required, particularly in UK towns and cities. CMAQ-urban used a kernel modelling technique, based upon the ADMS-Roads model, to describe the initial dispersion from every road source. The contribution from each source is then summed onto a fixed 20 m x 20 m grid close to roads, assuming that one can calculate the contribution of any source to total air pollution concentrations by applying each kernel and adjusting for the source strength. The kernels have been produced using an emissions source of unity (1 g km^{-1} s^{-1}) and using hourly meteorological data from the WRF model. A highly detailed treatment of road sources was required, with road emissions represented as a series of road links 10 m long and based on geographically accurate Ordnance Survey road map data. The streets classification included open roads (e.g.

motorways), typical roads (average urban roads surrounded by low rise buildings) and street canyons (classified by their orientation).

4.5. Industrial source emissions

All of the UK's point sources were included individually in the CMAQ-Urban model. These were typically large industrial and energy production facilities, such as oil refineries and power stations. Important release characteristics, such as stack height, plume velocity and temperature, have been incorporated into the model through the SMOKE System (Houyoux *et al* 2000). This ensures that the pollutant emissions have been released into the atmosphere at a height which accounts for the stack height plus plume rise.

5. London air pollution modelling methods

The London air pollution model also used a kernel modelling approach to describe the dispersion from each source and was based upon the ADMS-Roads and ADMS 5 models, using hourly WRF meteorological data. Since the London model represents every source type, a range of source kernels were used, including for industrial point sources, jet sources, volume sources and road and railway sources. The regional contribution of air pollution from outside of London in 2018 and 2030 was taken from the CMAQ-urban model. The specific treatment of major emissions sources in London was as follows:

5.1. Road sources

Within 500 m of a road, where strong concentration gradients exist, a highly detailed treatment of road sources is required, with emissions represented as a series of road links 10 m long and based on geographically accurate Ordnance Survey road map data. There were approximately 2.25 million 10 m road sources in London including open roads (motorways), typical roads (average urban roads surrounded by low rise buildings) and street canyons (classified by their orientation), with over 200 street canyon types.

5.2. Railway sources

Railway sources were treated in much the same way as for roads, i.e. by using the rail network emissions broken into 10 m sections. However, for diesel trains, the emissions release height was taken to be 5 m.

5.3. Part A industrial processes

Model kernels representing the varied release conditions (height, temperature, volume flow rate) were used for each part A process. A highly detailed treatment of these sources was required within 6km of each stack, to capture the maximum ground level plume concentrations.

5.4. Gas combustion sources

Gas combustion is a very important source of NO_X in London, and so a detailed representation of the height of release and spatial distribution of gas sources, as well as the temporal change in emissions throughout the year was included. Through analysis of the 3D model of buildings in London, the height of release from gas sources was varied from 1 m (domestic housing), through 25 m for small commercial premises to 75 m for large commercial office buildings. Gas heating sources were represented spatially by points located at 50 m intervals throughout the minor road network and set back from the road by 20 m. The model kernels used represented the varied release conditions from these sources, as well as a detailed treatment of the emissions variation by hour of the day and month of the year, taken from UK gas use statistics.

5.5. Heathrow airport sources

At Heathrow airport, emissions from aircraft during approach, landing, taxi out, taxi in, hold, take off, initial climb and climb out were represented as individual sources 10 m apart. Auxiliary Power Unit (APU) emissions and engine testing at the airport were represented as stationary point sources. Other sources such as heating plant, public and staff car parks, car rental, taxis queues and fire training ground emissions were represented horizontally as volume sources of 1 x 1 km, 50 m high, and for airside vehicles 2 m high.

Take off was the aircraft mode that provided one of the largest contributions to ground-level NO_X and PM concentrations. During take off each accelerating aircraft engine was represented by horizontal stationary jet sources whilst accounting for plume buoyancy. The hourly variation of aircraft emissions was reproduced using aircraft movements made available by the UK Civil Aviation Authority. Jet velocities were varied for the different aircraft operational settings of take off, approach and taxiing and were assumed to be 85%, 30% and 7% of full thrust, respectively. Finally, account was taken of the rapidly reducing effect of aircraft emissions on ground level concentrations at different aircraft heights.

5.6. Shipping sources

A detailed representation of the release height and spatial distribution of shipping sources as well as the temporal change in emissions have been included in the London model. Using four aggregate vessel categories, the height of release from shipping sources was varied from 5m (passenger vessels), through to 17.5m (fishing and tug) to 30m (bulk carrier and general dry cargo) and finally to 50m (tankers, containers, cruise ships etc.). In addition, a detailed treatment of the emissions variation by hour of the day and weekday/weekend was included in the model. Finally, vessel emission sources were represented spatially by points located every 20m and model dispersion kernels applied to these emissions to calculate their individual contribution to air pollution across London.

5.7. Other sources

All other sources, including domestic wood burning and cooking emissions in London, were represented as 1x1km emissions, mixed into a volume source with a height of 50m.

5.8. Predicting annual mean NO_2 concentrations

The method for converting NO_X to NO_2 used the well established relationships of Carslaw et al. (2001) and was based upon an analysis of measurements at both background and roadside sites. The conversion of NO_X to NO_2 also included the influence of NO_2 emitted directly from the vehicle exhaust.

6. Air Quality Model evaluation

6.1. Previous model evaluation

Imperial's novel CMAQ-urban model has been used widely for both health research (e.g. Smith *et al* 2016, Newbury *et al* 2019) and policy applications (e.g. Williams *et al* 2018a,b). It has undergone comprehensive evaluation as part of the recent UK DEFRA's model inter-comparison exercise[30] and internationally, as part of the AQMEII project (Solazzo *et al* 2017). Its ability to forecast future years has been demonstrated in a modelling study of pathway options to meet the 2050 UK Climate Change Act target and impacts on public health. The report and findings are published in the NIHR journal library (Williams *et al* 2018a) and Lancet Planetary Health (Williams *et al* 2018b). The model has also been used for UK compliance with $PM_{2.5}$ WHO guidelines in 2030 for DEFRA[31]. The modelling outcomes were reviewed by AQEG and published in 2019[32].

Not only is it important to establish whether the model can predict $PM_{2.5}$ across the UK in the base year 2018, but also that the performance of the model is consistent for other years, so long that it is run in similar ways and uses similar inputs, such as the NAEI. It is also a test of alternative meteorological years which are an important factor in 2030 forecasts. Previous (2012) published work for DEFRA are potentially informative and so model evaluation results for 2012 and 2018 have been given below, starting with 2018, the year on which the 2030 forecasts in this report are based.

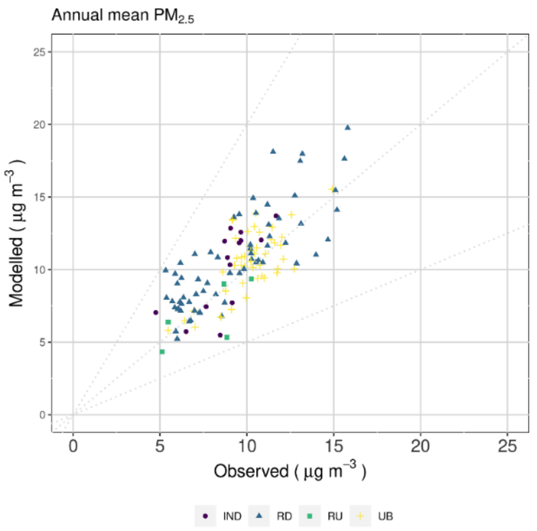
6.2. UK $PM_{2.5}$ model evaluation in 2018

The model evaluation in 2018 for $PM_{2.5}$ (see Figure 2), showed good agreement against 135 high quality UK fixed site monitors from rural to kerbside locations. In summary, all prediction were within a factor of two of the measurement (FAC2), the mean bias (MB) was 0.95 $\mu g\ m^{-3}$ (10%), a slight over prediction and the mean gross error (MGE) was 1.7 $\mu g\ m^{-3}$ (18%), showing scatter of the data to be more pronounced at roadside locations than at urban background sites. Overall this led to a good correlation coefficient (r) of 0.76. Finally, across a range of $PM_{2.5}$ components from the UK AGANET sites, the model performed reasonably well, with good agreement with ammonium aerosol,slight over predictions of sulphate and nitrate and an under prediction of chloride and sodium aerosols.

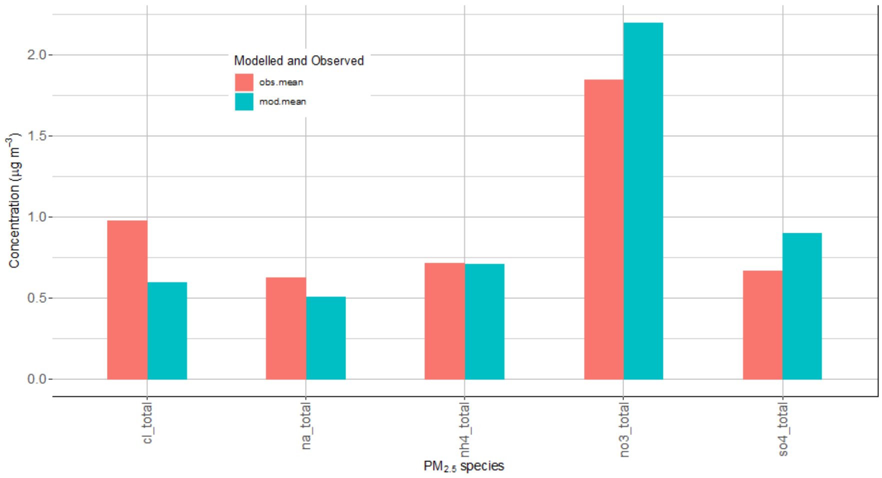
[30] https://uk-air.defra.gov.uk/research/air-quality-modelling?view=intercomparison (Accessed 09 February 2022).

[31] https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/930113/annex2-pm25-kings-college-report.pdf (Accessed 09 February 2022).

[32] https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/930104/air-quality-who-pm25-report.pdf (Accessed 09 February 2022).



Metric	$PM_{2.5}$
Number of sites	128
Observed mean (µg m^{-3})	9.5
Modelled mean (µg m^{-3})	10.4
FAC2	1
MB (µg m^{-3})	0.95
MGE (µg m^{-3})	1.7
NMB	0.1
NMGE	0.18
RMSE (µg m^{-3})	2.1
r	0.76

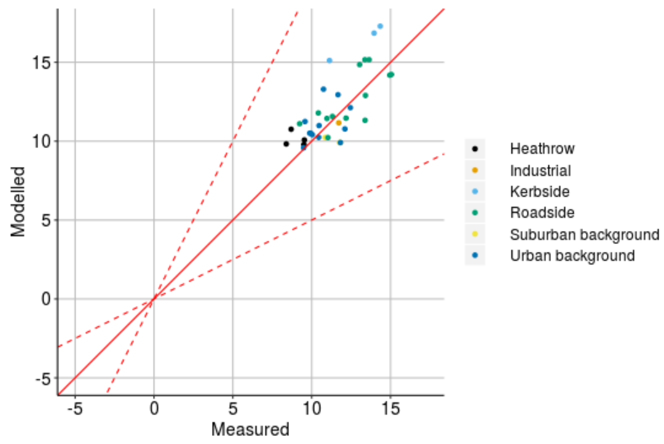


$PM_{2.5}$ species comparison against the UK Acid Gas Network (AGANET)

Figure 2 2018 model vs. observed estimates of $PM_{2.5}$ concentrations across all UK sites (left panel) and model performance statistics (right panel). Comparison of PM components (bottom panel)

6.3. London $PM_{2.5}$ model evaluation in 2018

The model evaluation in 2018 for $PM_{2.5}$ (see Figure 3), also showed good agreement against 26 high quality London fixed site monitors from suburban to kerbside locations. In summary, all predictions were well within a factor of two of the measurements (FAC2), the mean bias (MB) was 0.8 µg m^{-3} (7%), a slight over prediction and the mean gross error (MGE) was 1.5 µg m^{-3} (13%), showing scatter of the data uniformly across all site types. Whilst the bias of the London model was smaller than for the UK model the correlation coefficient (r) of 0.69 was smaller but still a good result.



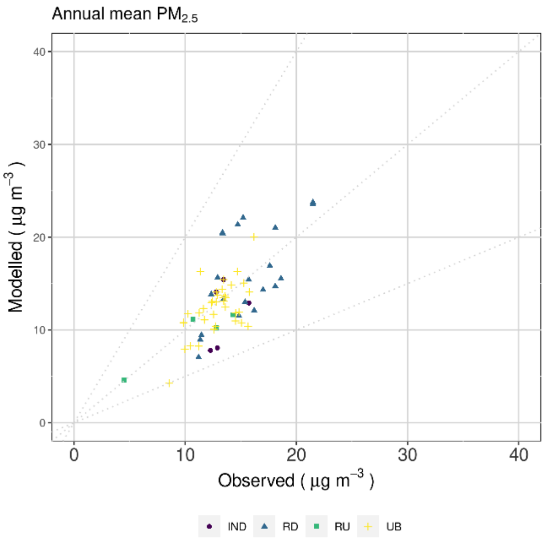
Metric	$PM_{2.5}$
Number of sites	26
Observed mean ($\mu g\ m^{-3}$)	11.7
Modelled mean ($\mu g\ m^{-3}$)	12.5
FAC2	1.0
MB ($\mu g\ m^{-3}$)	0.8
MGE ($\mu g\ m^{-3}$)	1.5
NMB	0.07
NMGE	0.13
RMSE ($\mu g\ m^{-3}$)	1.85
r	0.69

Figure 3 2018 model vs. observed estimates of $PM_{2.5}$ concentrations across all London sites (left panel) and model performance statistics (right panel)

6.4. UK $PM_{2.5}$ model evaluation in 2012

Similar $PM_{2.5}$ forecast modelling was undertaken for DEFRA and it is therefore worth comparing the 2012 model evaluation used in that project, with 2018 above, although note that the comparison can only be made for the CMAQ-urban model, not the London model, and also that in 2012 a smaller number of monitoring sites existed.

The model evaluation in 2012 for $PM_{2.5}$ (see Figure 4) showed good agreement against 86 high quality UK fixed site monitors from rural to kerbside locations. All of the predictions were within a factor of two of the measurement (FAC2), the mean bias (MB) was small, 0.4 $\mu g\ m^{-3}$ (3%), a slight under prediction and the root mean square error (RMSE) was 2.8 $\mu g\ m^{-3}$, showing scatter of the data across all sites. Overall this led to a reasonable correlation coefficient (r) of 0.66. Across a range of $PM_{2.5}$ components, the model performed well, with slight under predictions of sulphate, nitrate, ammonium and organic aerosols. Overall, these results demonstrate that whilst the model in 2012 had less bias and slightly more scatter, an indicator of uncertainty, the model performance between 2012 and 2018 was comparable.



Metric	$PM_{2.5}$
Number of sites	86
Observed mean (µg m^{-3})	13.0
Modelled mean (µg m^{-3})	12.6
FAC2	1.00
MB (µg m^{-3})	-0.4
MGE (µg m^{-3})	
NMB	-0.03
NMGE	
RMSE (µg m^{-3})	2.80
r	0.66

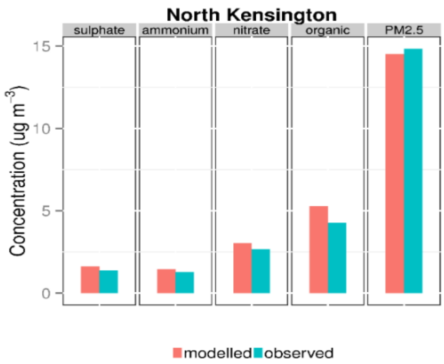


Figure 4 2012 model vs. observed estimates of $PM_{2.5}$ concentrations across all UK sites (left panel) and model performance statistics (right panel). Comparison of PM components (bottom panel)

6.5. UK and London NO_2 model evaluation in 2018

Whilst NO_2 predictions were not the focus of this project, they have been used in the health analysis, and are of widespread interest in the UK. We have therefore included UK maps for both 2018 and 2030. The most striking result is that whilst in 2018 (Figure 5 left panel) there was clear evidence of high concentrations in major cities and close to roads, some in excess of the EU 40 µg m^{-3} limit value, the considerable improvement in emissions performance brought about by both the newest Euro 6/VI vehicles, combined with widespread electrification of the UK vehicle fleet under the CCC BNZP, means that roadside and city centre concentrations have reduced considerably. Despite this improvement the new WHO 10 µg m^{-3} guideline value is still widely exceeded.

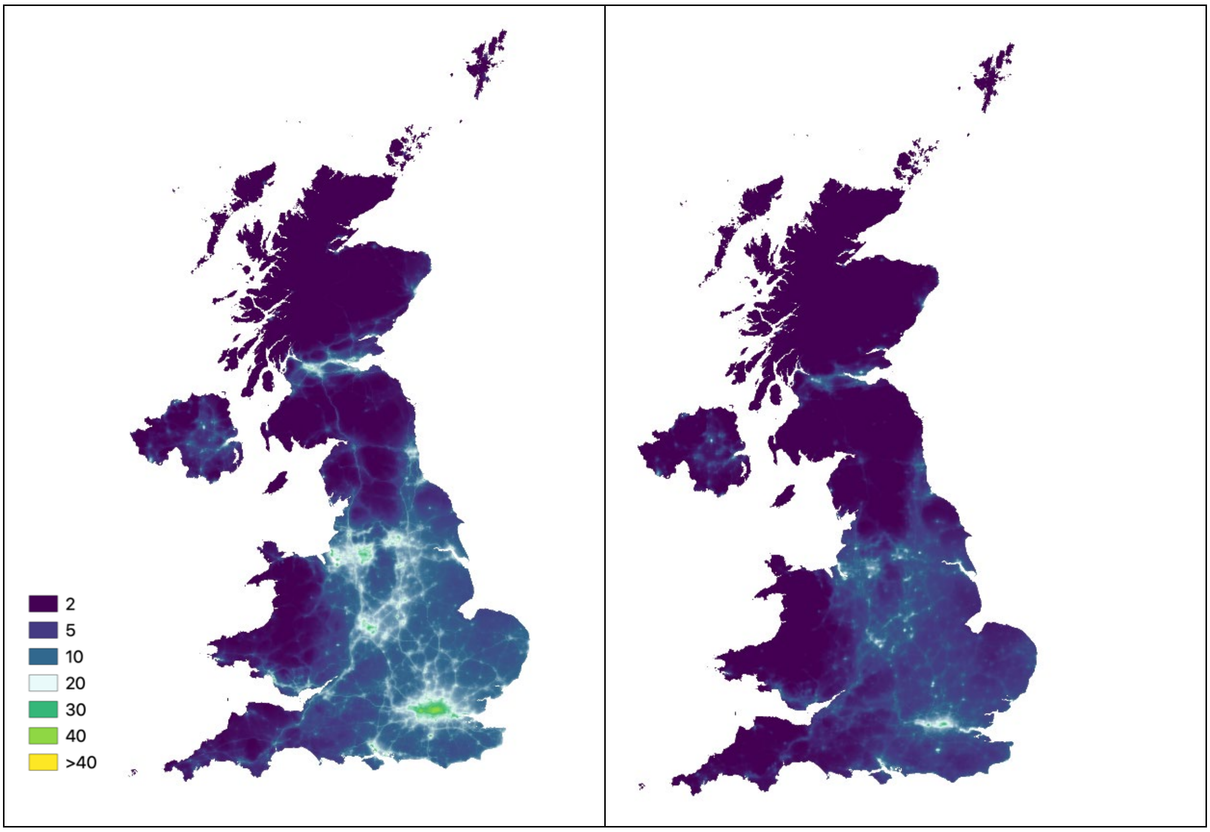
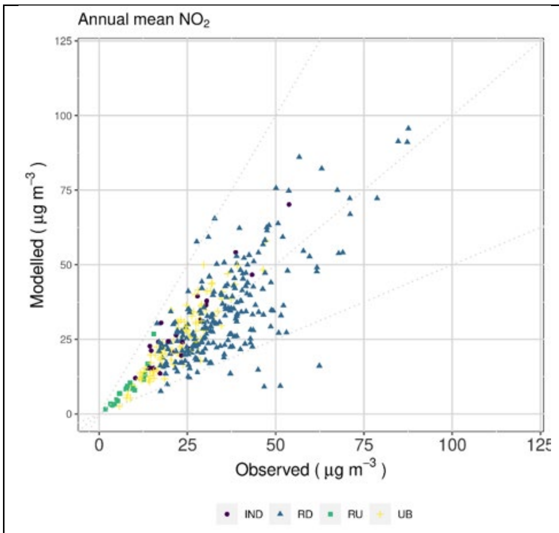


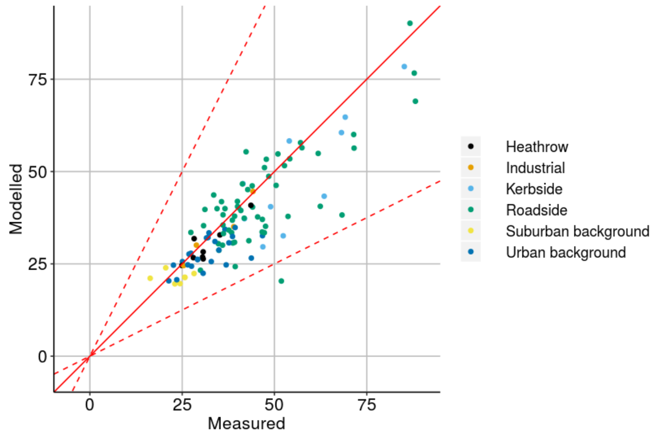
Figure 5 Forecasts of NO_2 concentrations in the UK in 2018 (left panel) and 2030 (right panel)



Metric	NO_2
Number of sites	370
Observed mean ($\mu g\ m^{-3}$)	29.9
Modelled mean ($\mu g\ m^{-3}$)	30.0
FAC2	0.97
MB ($\mu g\ m^{-3}$)	0.07
MGE ($\mu g\ m^{-3}$)	7.0
NMB	0.002
NMGE	0.23
RMSE ($\mu g\ m^{-3}$)	10.5
r	0.79

Figure 6 2018 model vs. observed estimates of NO_2 concentrations across all UK sites (left panel) and model performance statistics (right panel)

The UK model evaluation in 2018 for NO_2 (see Figure 6), showed good agreement against 370 UK high quality fixed site monitors from rural to kerbside locations. 97% of the predictions were within a factor of two of the measurement, the mean bias (MB) was virtually zero, and the mean gross error (MGE) was 7 µg m^{-3} (23%), showing scatter of the data principally at roadside sites. Some individual roadside sites were well under predicted, but this reflected locations where there was no road traffic data for that site location or where road traffic flows were themselves underpredicted. Overall the correlation coefficient (r) of 0.79 represents good agreement with UK measurements.



Metric	NO_2
Number of sites	109
Observed mean (µg m^{-3})	41.3
Modelled mean (µg m^{-3})	37.3
FAC2	0.99
MB (µg m^{-3})	-4.0
MGE (µg m^{-3})	6.2
NMB	-0.1
NMGE	0.15
RMSE (µg m^{-3})	8.62
r	0.86

Figure 7 2018 model vs. observed estimates of NO_2 concentrations across all London sites (left panel) and model performance statistics (right panel)

The London NO_2 model also performed well against 109 high quality fixed site monitors from suburban to kerbside locations (see Figure 7). 99% of the predictions were within a factor of two of the measurement, the normalised mean bias (MB) was -10%, and the mean gross error (MGE) was 6.2 µg m^{-3} (15%), showing scatter of the data across all site types. Overall the correlation coefficient (r) of 0.86 represented very good agreement with London measurements.

7. Health Impact Assessment methods

7.1. Health impact assessment approach

Health impact assessment takes results of epidemiological studies of associations between air pollutants and health outcomes and applies the response relationships from these studies to the predicted health impacts of policies to reduce air pollution. Four inputs are required –

- a modelled concentration change,
- a concentration-response relationship for change in the risk of the relevant health outcome per unit pollution concentration,
- the baseline rate for the health outcome per unit population and
- the population size of the population at risk.

The sections below give details of the inputs used and the method of calculation to derive the health impacts.

7.2. Design of health impact calculations

The details of the policies are described in section 2.2 This section describes the aspects relevant to the health benefits analysis.

Gains in life years as a result of reduced mortality is the major health benefit from reductions in air pollution. However, to predict these gains accurately it is necessary to run the life tables forward for an extended period. This is because changes in life years cannot be calculated until the predicted mortality as a result of the pollution changes has occurred. For many in the population that will not be for many decades. We therefore chose to set up the lifetable calculations assuming that the reduced levels of pollution achieved in 2030 were maintained for a lifetime (105 years beyond that i.e. to 2134) (Figure 8).

As the lifetable calculations were set up to cover an extended period, the same was done for the other health outcomes. These were calculated as both totals and averages per year. This does not mean the results were the same each year because the evolving population size and age distribution as predicted by the lifetables was used as a different input each year.

For the UK life years calculations, comparisons were made between the UK 2030 plus LS1 scenario and assuming 2018 levels of pollution remained unchanged (the counter-factual). The remaining scenarios were calculated on a London basis (Figure 8) with comparison with either the 2018 counterfactual or the UK2030 plus LS1 scenario in London.

For the other health outcomes, which were calculated in a more approximate way, all scenarios were calculated on a UK basis compared with the 2018 concentrations remaining unchanged. The comparisons between scenarios UK2030+LS2 and UK2030+LS3 and scenario UK2030+LS1 were equivalent to the impact in London because the concentration changes only occurred in London.

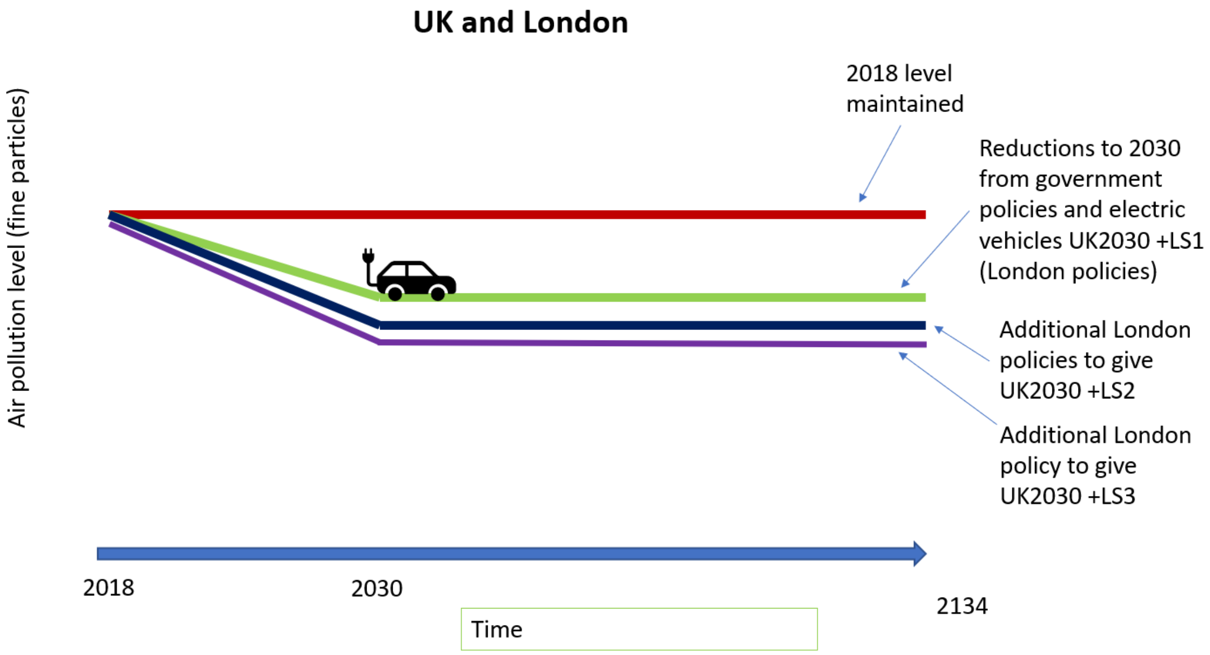


Figure 8 Schematic of the comparison between the UK and the London scenarios for life-table calculations and calculations for other health outcomes

7.3. Air Quality data

From 20m grid data to ward concentration

Particulate matter with diameter <2.5 µm ($PM_{2.5}$) and nitrogen dioxide (NO_2) annual mean concentrations across the UK and London were predicted for the years 2018 and 2030 and the air pollution data was intersected with the latest Ward layer from the Office of National Statistics (ONS). Each concentration grid point within each Ward was further averaged at local authority level (LA), weighting by ward level population.

Anthropogenic $PM_{2.5}$: Non-anthropogenic $PM_{2.5}$ was derived by Ward using CMAQ data for the years 2018 and 2030 and subsequently by subtracting the modelled contribution from natural aerosols sources such as sea-salt - from the total $PM_{2.5}$ modelled to generate anthropogenic $PM_{2.5}$ concentrations; consistent with EU guidance (European Commission, 2011[33]).

From ward to population-weighted LA concentration

Population-weighted average concentration (PWAC): Population-weighting was summarised at local authority level. To do this the Ward averaged concentrations were multiplied by the population aged 30 plus for each gender and the resulting population-concentration product summed across all Wards in each LA and then divided by the LA population. The LA population-weighted means were then used directly in the life table calculations.

[33]https://ec.europa.eu/environment/air/quality/legislation/pdf/sec_2011_0208.pdf (Accessed 09 February 2022).

From ward concentration to UK average concentration (not population-weighted)

The main health benefits analysis concentrated on the life years gained – the largest of the hbenefits. Some additional morbidity outcomes were also calculated using simpler approaches due to time constraints and to limited availability of baseline health data at a local level. These calculations were done at UK level only and used the average of the ward concentrations across the UK rather than population-weighted concentrations. Unlike for the life years calculations, the calculations used the difference between the 2018 baseline and the 2030 concentrations for each scenario throughout the time period, rather than interpolation between 2018 and 2030. Average concentrations are similar or slightly lower than population-weighted average concentrations (see results).

7.4. Health evidence – concentration-response functions, baseline rates and calculation methods

The following sections are divided by health outcome. Within each section, the population at risk (e.g. whole population, asthmatics), the concentration-response function and the baseline rates (the typical number of health outcomes occurring in the relevant population, irrespective of changes in air pollution) are set out.

Calculation methods are set out where needed. This is not in every section as the calculation methods are the same for several health outcomes. The method differs according to the method of analysis in the underlying epidemiological studies i.e. Cox proportional hazards model for time to event e.g. life years, Poisson regression for rare events that occur as counts e.g. hospital admissions and logistic regression for health outcomes analysed as present or absent e.g. symptoms.

It is now well established that adverse health effects, including mortality, are statistically associated with outdoor ambient concentrations of air pollutants. Moreover, toxicological studies of potential mechanisms of damage have added to the evidence such that many organisations (e.g. US Environmental Protection Agency[34]; World Health Organization[35], COMEAP[36]) consider the evidence strong enough to infer a causal relationship between the adverse health effects and the air pollution concentrations. Causality aspects have been taken into consideration and are discussed in the relevant sections where health outcomes are not already well established

7.5. Long-term exposure to $PM_{2.5}$ and NO_2 and all-cause mortality

The 2018 Committee on the Medical Effects of Air Pollutants (COMEAP) report includes two options for concentration-response functions for use in impact calculations according to whether the analysis is for a policy or mixture of policies that reduces air pollution (NO_2 and $PM_{2.5}$) as a whole or is for a NO_2 specific policy. We considered that the former was more

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[34]https://www.epa.gov/isa (Accessed 09 February 2022).

[35]Review of evidence on health aspects of air pollution (REVIHAAP) Available at: (http://www.euro.who.int/__data/assets/pdf_file/0020/182432/e96762-final.pdf (Accessed 09 February 2022).

[36]https://www.epa.gov/isa (Accessed 09 February 2022).

appropriate since the range of policies considered in this study are aimed at $PM_{2.5}$ but also reduce NO_2.

A full health impact assessment requires a follow-up of the initial population for a life-time even if the pollution changes are only for the next decade or so. In this study, the health benefits of pollution changes over the period 2018-2030 have been calculated using the full result for gains in life expectancy until everyone in the initial population has died by 2134 (i.e. 105 years from 2030).

Concentration-response functions

The concentration-response functions used and the spatial scales of the input data is given in Table 10. The concentration-response functions are based on the latest advice from COMEAP in 2018 (COMEAP, 2018[37]) for NO_2 with $PM_{2.5}$ aspects updated using the meta-analysis by Chen and Hoek (2020) as discussed by COMEAP at their March 2021 meeting (COMEAP, 2021b)[38]. Results are given without a cut-off for $PM_{2.5}$ and, with and without a cut-off of 5 µg m^{-3} for NO_2.

Table 10 Concentration-response functions (CRFs) for long-term exposures and mortality (for impact calculations of general changes in pollutant concentrations (rather than policies targeting one pollutant alone)

Pollutant	Averaging time	Population at risk	Hazard ratio per 10 µg m^{-3}	Baseline rate	Cut-off	Lag
$PM_{2.5}$	Annual average	Adults age 30+ (2019 data)	HR 1.08 (1.06 – 1.09) (Chen and Hoek, 2020)	Age-specific mortality by gender, single year of age (see text for further details)	Zero	US EPA lag
NO_2			HR 1.023 (1.008-1.037) (COMEAP, 2018a[39])		Zero or 5 µg m^{-3}	US EPA lag

HR Hazard ratio – the ratio between hazard rates (age-specific mortality rates)

[37]COMEAP (2018) Associations of long-term average concentrations of nitrogen dioxide with mortality. A report by the Committee on the Medical Effects of Air Pollutants. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/734799/COMEAP_NO2_Report.pdf (Accessed 09 February 2022).

[38] See www.comeap.org.uk for minutes of COMEAP meetings.

[39]COMEAP (2018) Associations of long-term average concentrations of nitrogen dioxide with mortality. A report by the Committee on the Medical Effects of Air Pollutants. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/734799/COMEAP_NO2_Report.pdf (Accessed 09 February 2022).

Lags

The approach allowed for a delay between exposure and effect using the recommended distribution of lags from COMEAP (COMEAP, 2010[40]) who agreed to adopt the US EPA lage i.e. 30% of the effect in the first year, 12.5% in each of years 2-5 and 20% spread over years 5-20. An analogous approach was used for the effects of long-term exposure to NO_2. HRAPIE (WHO, 2013[41]) recommended that, in the absence of information on likely lags between long-term exposure to NO_2 and mortality, calculations should follow whatever lags are chosen for $PM_{2.5}$.

Population, death, mortality improvements and birth projections inputs

Population data

Population data in England and Wales: the population data for the year 2019[42] has been obtained from ONS by gender and by single year of age at Ward level[43].

Population data in Scotland: the population data for the year 2019 has been obtained from National Records of Scotland by gender and by single year of age at Ward level[44].

Population data in Northern Ireland: the population data for the year 2019 has been obtained from NISRA by gender and by broad age bands (0-15, 16-39, 40-64 and 65+) at Ward level[45]. The population was further redistributed by gender and by single year of age at Ward level using data for the year 2019 obtained from NISRA by gender and by single year of age at Administrative Areas[46].

The 2019 population data was used subsequenbtly to represent the life table population in 2018.

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[40]COMEAP 2010, The mortality effects of long-term exposure to particulate matter air pollution in the UK, London, UK. Available at http://comeap.org.uk/documents/reports/128-the-mortality-effects-of-long-term-exposure-to-particulate-air-pollution-in-the-uk.html (Accessed 09 February 2022).

[41]WHO (2013), Health risks of air pollution in Europe-HRAPIE project, WHO Regional office for Europe. Available at: http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/publications/2013/health-risks-of-air-pollution-in-europe-hrapie-project.-recommendations-for-concentrationresponse-functions-for-costbenefit-analysis-of-particulate-matter,-ozone-and-nitrogen-dioxide (Accessed 09 February 2022).

[42]For Wards, we obtained the Wards shapefile version 2019 and as a result, we had to use 2019 wards data for population data. In other cases (such as death, birth and mortality improvement), we could be more flexible (and use a combination of year or 2018) as data required was at local authority.

[43]https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationestimates/datasets/wardlevelmidyearpopulationestimatesexperimental (Accessed 09 February 2022).

[44]https://www.nrscotland.gov.uk/statistics-and-data/statistics/statistics-by-theme/population/population-estimates/2011-based-special-area-population-estimates/electoral-ward-population-estimates (Accessed 09 February 2022).

[45]https://www.nisra.gov.uk/publications/2019-mid-year-population-estimates-northern-ireland (Accessed 09 February 2022).

[46]https://www.nisra.gov.uk/publications/2019-mid-year-population-estimates-northern-ireland (Accessed 09 February 2022).

Deaths data

Deaths data in England and Wales: the death data has been obtained from ONS[47] by gender and by single year of age at LSOA level and averaged for the years 2016/2017/2018 to represent 2018[48]. LSOA level deaths data were available for the year 2016[49] and requested directly from ONS for the years 2017-2018. The deaths data was further aggregated by gender and by single year of age at local authority level using the LSOA data as above.*Deaths data in Scotland:* the deaths data has been obtained from Statistics.sco.gov[50] by gender and by single year of age at Council area level[51] and averaged for the years 2016/2017/2018 to represent 2018.

Death data in Northern Ireland: the death data has been obtained directly from NISRA[52] by gender and by single year of age at local authority level and averaged for the years 2016/2017/2018 to represent 2018.

Note that deaths data for subsequent years were projected within the life-tables. This means that it does not take into account of the increased mortality from COVID-19 in 2020. We considered that any analysis to take this into account was best done after the pandemic when a full update could be completed.

Birth Projections 2018 – 2134

Projections of the total number of births per local authority were derived for England, Wales, Scotland and Northern Ireland for both males and females.

Changes in births over time –

- Actual data on numbers of births by gender at Wards level was used in 2018 then aggregated up to local authority level for England, Wales, Scotland and Northern Ireland (using Ward population data as descibed above)

[47]https://www.ons.gov.uk/ (Accessed 09 February 2022).

[48] 3 year averages are often used to give a more robust baseline avoiding the influence of atypical years. We has the years 2016/17/18 from a previous project so used this rather than 2017/18/19.

[49]https://www.ons.gov.uk/peoplepopulationandcommunity/birthsdeathsandmarriages/deaths/adhocs/007713deathsbylowersuperoutputareasexandsingleyearofageenglandandwales2016 (Accessed 09 February 2022).

[50]https://statistics.gov.scot/home (Accessed 09 February 2022).

[51]https://statistics.gov.scot/resource?uri=http%3A%2F%2Fstatistics.gov.scot%2Fdata%2Fdeaths (Accessed 09 February 2022).

[52]https://www.nisra.gov.uk/ (Accessed 09 February 2022).

- Birth projections by local authority (2018 based edition for England[53], Wales[54] and Northern Ireland[55]) and council areas (2018 based edition for Scotland[56]) were obtained by gender from 2019 to 2043
- The year-on-year projected percentage change in births estimated by ONS for Great Britain[57] (PPP - principal projection) was then applied to scale 2043 births for each year from 2044 up until 2118 for each local authority (and for each gender)
- No projections were available after 2118 so births were left constant for each local authority by gender for all years from 2119 to 2134

Mortality improvements 2018 – 2134

Changes in mortality rate improvements over time –

- Mortality rate improvements obtained from ONS (2018[58] based edition) by gender and age were used in England, Wales, and Northern Ireland (UK excluding Scotland data) and Scotland (Scotland data separately) for all years from 2019 to 2134
- Note that the rate of mortality improvement was left constant for the period 2044 to 2134

Migration

Predicting migration at the current time post the European referendum is particularly uncertain with both increases and decreases forecast. We did not therefore include this in our first analyses as presented in this report. Over the country, as a whole, this contribution to overall health impacts is likely to be small. This can be explored further in future work.

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[53] https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationprojections/datasets/localauthoritiesinenglandz1 (Accessed 09 February 2022).

[54] https://statswales.gov.wales/Catalogue/Population-and-Migration/Population/Projections/Local-Authority/2018-based/populationprojections-by-localauthority-year (Accessed 09 February 2022).

[55] https://www.nisra.gov.uk/publications/2018-based-population-projections-areas-within-northern-ireland (Accessed 09 February 2022).

[56] https://www.nrscotland.gov.uk/statistics-and-data/statistics/statistics-by-theme/population/population-projections/sub-national-population-projections/2018-based/detailed-datasets (Accessed 09 February 2022).

[57] https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationprojections/datasets/z2zippedpopulationprojectionsdatafilesgbandenglandandwales (Accessed 09 February 2022).

[58] https://www.ons.gov.uk/peoplepopulationandcommunity/birthsdeathsandmarriages/lifeexpectancies/adhocs/11827calendaryearmortalityimprovementsfor2018basedprojectionsukexcludingscotlandandscotlandseparately (Accessed 09 February 2022).

Mortality impact calculations

The relative risk (RR) per 10 µg m^{-3} was scaled to a new relative risk for the appropriate population-weighted mean for each gender in each ward for each scenario and year. The equation used (for the example coefficient of 1.08) was: RR(x) = 1.08x/10 where x is the concentration change of interest (with a negative sign for a reduction). Concentrations were assumed to reduce linearly between the years in which modelled concentrations were available (2018 and 2030). The scaled RR was then used to adjust the all-cause hazard rates in the life table calculations.

For the 5 µg m^{-3} cut-off for NO_2, local authority concentrations were interpolated between the years in which modelled concentrations were available (2018 and 2030) and 5 µg m^{-3} was then subtracted from the ward concentrations in each year. Any resulting negative concentrations were then set to zero before all the ward concentrations were population-weighted to local authority level.

Life table calculations were programmed in SQL based on the methods used in the standard IOMLIFET spreadsheets with the following amendments:

- Extension to 2134 (105 years after 2030)
- Adjustment of the baseline hazard rates over time according to projected mortality rate improvements
- Inclusion of changes in numbers of births over time
- IOMLIFET excludes neonatal deaths. We included neonatal deaths and followed the South East Public Health Observatory life-expectancy calculator[59] and Gowers et al. (2014) in taking into account the uneven distribution of deaths over the course of the first year when calculating the survival probability. (The survival probability (the ratio of the number alive at the end of the year to the number alive at the beginning) is derived by the equivalent of adding half the deaths back onto the mid-year population to give the starting population and subtracting half the deaths from the mid-year population to give the end population, assuming deaths are distributed evenly across the year. This is not the case in the first year where a weighting factor based on 90% of the deaths occurring in the first half of the year and 10% in the second half is used instead. After rearrangement the actual formula is (1- 0.1 x hazard rate)/(1+ 0.9 x hazard rate) rather than the (1- 0.5 x hazard rate)/(1+ 0.5 x hazard rate) used in other years.

Local authority/country output: The changes in life years in the life tables were then summed across the total population and the full time period in each local authority. Results for total and annual life years lost by local authority were then summed to Greater London, Greater Manchester and country level. We also used the life tables to calculate changes in life expectancy and to contribute changing population data over time for the calculation of other health outcomes.

[59] https://webarchive.nationalarchives.gov.uk/20130329125326/http://www.lho.org.uk/viewResource.aspx?id=8943&sUri=http%3a%2f%2fwww.sepho.org.uk%2f

7.6. Other health outcomes

Post-neonatal all-cause mortality (infant deaths) (short-term exposure) PM_{10}

Post-neonatal all-cause mortality refers to deaths in infants aged 1-12 months (deaths at less than 1 month are less likely to have a contribution from environmental factors so are excluded). Usually, the short-term exposure and mortality concentration response functions are not included because they overlap with the studies of long-term exposure to pollutants and mortality. However, the latter is only applied to the adult population age 30+. The evidence on air pollution and post-neonatal mortality does not overlap so this can be included. There is a concentration-response function for post-neonatal all-cause mortality and PM_{10} recommended by WHO (WHO, 2013[60]). Ideally, this would be included within the life-table analysis described above but this would have required more time for methodological development than was possible in this project. Instead, this was calculated separately. For simplicity, we used all births as the population at risk and all deaths in the first year. This will result in an overestimate but the impact on the overall results will be small as it is a rare outcome.

There are some uncertainties associated with this outcome – the WHO recommendation was based on a relatively old study (Table 11) and it is an endpoint that is not studied very often. Fortunately, deaths in infants aged 1-12 months are rare meaning that large studies would be needed to detect any effects. As the numbers are small but the monetary valuation is large, this can lead to marked swings in monetised benefits for small changes in assumptions.

Table 11 Calculation inputs for air pollution and post-neonatal mortality

Population at risk	Baseline rate	Concentration-response function
Total infants aged 0 -12 months in the UK for the years 2018-2134 from the births data used in the lifetable analysis (incorporating birth projections)	Infant death rates from ONS[61] for Englan and Wales	Odds ratio 1.04 (1.02, 1.07) per 10 μg m^{-3} (Woodruff et al 1997)

The calculation method is set out in the section on chronic bronchitis/phlegm.

[60]WHO (2013), Health risks of air pollution in Europe-HRAPIE project, WHO Regional office for Europe. Available at: http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/publications/2013/health-risks-of-air-pollution-in-europe-hrapie-project.-recommendations-for-concentrationresponse-functions-for-costbenefit-analysis-of-particulate-matter,-ozone-and-nitrogen-dioxide (Accessed 09 February 2022).

[61]https://www.ons.gov.uk/peoplepopulationandcommunity/birthsdeathsandmarriages/deaths/bulletins/childhoodinfantandperinatalmortalityinenglandandwales/2019 (Accessed 09 February 2022).

Coronary heart disease $PM_{2.5}$

COMEAP has reviewed the effects of air pollution on cardiovascular morbidity. While the full report is not yet published, the concentration-response function is quoted in the COMEAP advice note on health evidence relevant to developing targets for $PM_{2.5}$ under the Environment Bill (COMEAP,2021[62]). The recommendation is for incidence (new cases) of ischaemic heart disease (a term used more or less interchangeably with coronary heart disease, the term we will use in this report,) One issue discussed in the COMEAP minutes[63] is whether the effect of air pollution on case fatality (deaths in those with coronary heart disease) is stronger than that for incidence (COMEAP, 2018b). If this is the case, then reducing air pollution could reduce new cases of coronary heart disease but also, to a greater degree, increase life expectancy in those with heart disease. This is a good thing but could result in an increase in the total numbers (prevalence) of people with heart disease at any one time. While studies on case fatality suggest this might be so, there are too few of them to be sure. We have not incorporated this aspect here – this might result in an overestimate of the number of cases reduced by reducing air pollution. We have offset this possibility to some extent in other choices (see section on stroke).

The inputs to the calculation are shown in Table 12. We used age 30+ for the population at risk – the available European cohorts in the meta-analysis by Cesaroni et al (2014) used a variety of lower age limits – 25,35,45 and older – so the choice of age range was not clear cut.

Table 12 Calculation inputs for incidence of coronary heart disease

Population at risk	Baseline rate	Concentration-response function
UK Total age 30+ for each year 2018 – 2134 Derived from single year of age population data generated by the life table calculations Prevalence cases of CHD (from BHF) were subtracted to give numbers of people without CHD	Derived from British Heart Foundation estimates for total UK cases for 2019 https://www.bhf.org.uk/what-we-do/our-research/heart-statistics/heart-statistics-publications/cardiovascular-disease-statistics-2021	COMEAP (2021a) 1.07 (95% CI 0.99, 1.16) per 10 µg m^{-3} increase in $PM_{2.5}$ for ischaemic (coronary) heart disease incidence

Ideally, this outcome would be analysed using time to event analysis, and linked in with the life-tables. However, there was insufficient time to develop this for this short project. Instead it was analysed on the basis of new cases each year, treating each year independently. For

62 https://www.gov.uk/government/publications/fine-particulate-air-pollution-pm25-setting-targets (Accessed 09 February 2022).

63 https://www.gov.uk/government/groups/committee-on-the-medical-effects-of-air-pollutants-comeap (Accessed 09 February 2022).

computational convenience we used the same method as for most of the other health outcomes, i.e. using odds ratios. This underestimated the answer that would have been obtained using a relative risk (see method for hospital admissions) by about 5%. This is also countered by the possible overestimation as a result of omitting the influence of a possible greater effect on case fatality.

Chronic bronchitis (chronic phlegm) PM_{10}

COMEAP published a report on air pollution and chronic bronchitis in 2016 (COMEAP, 2016). The studies they examined were intended to examine whether the incidence of chronic bronchitis as a permanent disease. Epidemiological studies often use questionnaires to define disease. In one of the key studies examined (Schindler et al 2009), it was found that some of those reporting chronic cough or phlegm[64] no longer reported it when followed up ten years later. This suggests the effect may be more reversible than first thought, and less indicative of a long-lasting and progressive disease. Reviewing the studies overall, COMEAP concluded that air pollution and chronic bronchitis should not be quantified in core analysis of health benefits.

COMEAP did, however, recommend a method for quantification for use in sensitivity analysis. This involved using an association with chronic phlegm from Cai et al (2014). We followed this method and inputs (Table 13) except that, rather than using it as a sensitivity analysis with a monetary valuation based on chronic bronchitis as a disease, we included it in core analysis but with a lower monetary valuation based on symptoms (*Table 20*).

Table 13 Calculation inputs for chronic bronchitis (chronic phlegm)

Population at risk adults **age 16+**	Baseline daily prevalence of chronic phlegm, non-smokers, 16+	Concentration-response function
UK Total age 16+ for each year 2018 – 2134. Derived from single year of age population data generated by the life table calculations.	5% England, Wales and Northern Ireland from 2010 Health Survey for England (NHS Digital, 2011). 4.6% Scotland from the 2010 Scottish Health Survey (Scottish Government, 2011) (equivalent to 4.9% UK) (assumed to apply in future years)	Odds ratio 1.32 (95% CI) 1.02, 1.71) per 10 µg m^{-3} increase in **PM_{10}** Cai *et al* (2014)

For this method, it is the odds rather than the risk that is adjusted to account for the concentration increment being analysed.

[64] *Chronic cough or phlegm* was defined as *chronic cough* and/or *chronic phlegm*, with “chronic” being defined by the presence of the respective symptoms during at least 3 months per year for at least two years.

Briefly, the baseline prevalence (probability - p) of chronic phlegm was converted to baseline odds for no change in $PM_{10}(Odds_{baseline})$ using the equation for no change in $PM_{10}(Odds_{baseline}) = p/(1-p)$.

Then we take the odds ratio (OR) per 10 μg m^{-3} increase in PM_{10} to work out the odds of a10 μg m^{-3} increase ($Odds_{10}$) using the equation $Odds_{10} = OR \times Odds_{baseline}$.

Then, we calculated the Odds per decrease in concentration for the relevant scenario on the log scale and backtransformed it into a probability. This probablility was then applied to the baseline rate and the population age 16+. This gives the air pollution attributable annual number of cases of chronic phlegm.

An analogous worked example is given in Annex 10 of Walton et al (2015)

Hospital admissions (short-term exposure) $PM_{2.5}$ and NO_2

We used concentration-response functions from published meta-analyses from a Department of Health funded systematic review (Atkinson et al 2014; Mills et al 2015). These were from single pollutant models so it is not appropriate to add the results together (see discussion in results section). Instead the largest result out of the two pollutants was used.

There was insufficient time for the permission processes involved in obtaining hospital episode statistics or the multiple calculations at a fine spatial scale. Instead baseline rates at national level were compiled from publicly available sources. The way in which these were compiled was different for the different constituent countries:

England Emergency admissions data from the hospital admitted patient care activity[65]: diagnosis file were summed across the relevant ICD codes using statistics for the whole of England.

Wales 2018/19 emergency admissions data[66] on Primary Diagnosis (3 character detail) by Local Health Board of Residence Welsh Resident was summed across the relevant ICD codes and across the local health boards to give figures for Wales for all respiratory and all cardiovascular admissions.

Scotland Admissions data from the relevant grouping of ICD codes was obtained by selecting Scotland in the Diagnosis by Council Area of Residence file[67] at Emergency admissions data was not provided so this was estimated by using the proportion of emergency admissions to total admissions data for England.

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[65] https://digital.nhs.uk/data-and-information/publications/statistical/hospital-admitted-patient-care-activity/2018-19 (Accessed 09 February 2022).

[66] https://nwis.nhs.wales/information-services/health-intelligence/annual-pedw-data-tables/ (Accessed 09 February 2022).

[67] https://www.isdscotland.org/health-topics/hospital-care/diagnoses/ (Accessed 09 February 2022).

Northern Ireland Emergency admissions data were summed across the relevant ICD codes using statistics for the whole of Northern Ireland[68].

The calculation method was that for concentration response functions from epidemiological studies using Poisson Regression. For this type of analysis, it is the log of the relative risk that is plotted against concentration. So, the percentage change per 10 µg m^{-3} from Table 14 is converted to a relative risk (dividing by 100 and adding 1). The natural log of this relative risk is then divided by 10 and multiplied by the relevant concentration increment for each pollutant and scenario. After converting back to a percentage change, this is applied to the baseline number of hospital admissions to give the change in air pollution attributable hospital admissions. For years subsequent to 2018, the calculation was repeated except that the baseline number of hospital admissions was calculated from the baseline rate for 2018 and the population for 2019 and future years from the lifetable calculations. The results were summed over the entire time period for use in the economic analysis and averaged to give the average cases per year.

Table 14 Calculation inputs for respiratory and cardiovascular admissions

Population at risk	Baseline rate of hospital admissions	Concentration-response function (% per 10 µg m^{-3})
UK Total all ages for each year 2018 – 2134 Derived from single year of age population data generated by the life table calculations	Emergency hospital admissions ICD 10 codes J00-J99 and I00-I99 from national statistics sources (see text)	Respiratory hospital admissions: 24-hour average $PM_{2.5}$ 0.96 (-0.63, 2.58) 24-hour average NO_2 0.57 (0.33, 0.82) Cardiovascular hospital admissions: 24-hour average $PM_{2.5}$ 0.90 (0.26, 1.53) 24-hour average NO_2 0.66 (0.32, 1.01) Sources: Atkinson et al (2014) for $PM_{2.5}$. Mills et al (2015) for NO_2.

[68] https://www.health-ni.gov.uk/publications/acute-episode-based-activity-downloadable-data-201819 (Accessed 09 February 2022).

Asthmatic symptoms in asthmatic children (PM_{10} short-term exposure)

This endpoint and inputs (Table 15) was based on recommendations from the HRAPIE project (WHO, 2013[69]), in turn based on Weinmayr et al 2010. The latter reference also included a pooled odds ratio for NO_2. This was not included in the WHO recommendations and we did not use it either because of possible overlap with the calculations for bronchitic symptoms in asthmatic children as a result of long-term exposure to NO_2, which we did quantify.

We considered potential overlap between quantifying different outcomes related to asthma. Asthma admissions are included within respiratory admissions, which are also quantified. However, the panel studies used to derive the CRF for asthmatic symptoms are unlikely to have included any hospital admissions as hospital admissions are fortunately much rarer than symptoms and are unlikely to occur in a panel of small numbers of children. We did not include asthma incidence (see below).

Table 15 Calculation inputs for asthmatic symptoms in asthmatic children

Population at risk step 1 children age 5-19	Population at risk step 2 asthmatic children age 5-19	Baseline daily prevalence of asthmatic symptoms in asthmatic children	Concentration-response function
UK Total age 5-19 for each year 2018 – 2134 Derived from single year of age population data generated by the life table calculations	Proportion of children with severe asthma from Lai et al 2009 (11.45%)	17% (from interpolation of several panel studies by HRAPIE (WHO, 2013[70]))	OR 1.028 (1.006–1.051) per 10 µg m^{-3} PM_{10} (Weinmayr et al 2010) Recommended by WHO (2013)

The calculation method is that used for studies based on logistic regression – see section on chronic bronchitis/phlegm. The only difference was that the population at risk (asthmatic children) had to be derived first, applying the proportion of children with severe asthma and the total number of children age 5-19.

[69] WHO (2013), Health risks of air pollution in Europe-HRAPIE project, WHO Regional office for Europe. Available at: http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/publications/2013/health-risks-of-air-pollution-in-europe-hrapie-project.-recommendations-for-concentrationresponse-functions-for-costbenefit-analysis-of-particulate-matter,-ozone-and-nitrogen-dioxide (Accessed 09 February 2022).

[70] WHO (2013), Health risks of air pollution in Europe-HRAPIE project, WHO Regional office for Europe. Available at: http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/publications/2013/health-risks-of-air-pollution-in-europe-hrapie-project.-recommendations-for-concentrationresponse-functions-for-costbenefit-analysis-of-particulate-matter,-ozone-and-nitrogen-dioxide (Accessed 09 February 2022).

Bronchitic symptoms in asthmatic children (NO_2 long-term exposure)

The WHO recommendation is to use the data for numbers of children age 13-14 with 'ever asthma' from Lai et al (2009) to determine the number of asthmatics (the original study used doctor diagnosed asthma) (Table 16). WHO specified numbers of 5.1 % for Northern and Eastern Europe; and 15.8% for Western Europe. However, Lai et al 2009 also has figures for the UK (25%) and for London (11.45%). These latter figures were used as a range 11.45% - 25% in the 2015 report.on health impacts of air pollution in London (Walton et al., 2015). We decided to use the value of 11.45% for this report as asthma rates have declined somewhat in recent years.

There are slightly more recent figures for 2012[71] but these do not distinguish between 'severe asthma' and 'ever asthma', as is done in Lai et al 2009. The figures are of a similar order though (around 12% across all ages, 15% age 11 to 15).

The calculation method is that used for studies based on logistic regression – see section on chronic bronchitis/phlegm. The population at risk (asthmatic children) had to be derived first.

Table 16 Calculation inputs for bronchitic symptoms in asthmatic children

Year	Population at risk part 1 children age 5-14	Population at risk part 2 asthmatic children age 5-14 (this will be the input to the calculations)	Baseline daily prevalence of bronchitic symptoms in asthmatic children	Concentration-response function
2019	UK Total age 5-14 for each year 2018 – 2134 Derived from single year of age population data generated by the life table calculations	Proportion of asthmatics 'ever asthma' age 13-14 from Lai et al 2009 11.45%	Prevalence of bronchitic symptoms among asthmatic children 21.1% to 38.7% (Migliore et al., 2009; McConnell et al., 2003) (Used 38.7% for this report)	OR 1.021 (0.99–1.06) per 1 $\mu g\ m^{-3}$

Acute bronchitis in children (PM_{10}, short-term exposure)

Acute bronchitis is a respiratory infection. It can last for up to 3 weeks, although the cough can go on a bit longer. It is usually treated at home with rest and anti-inflammatories. If

[71] https://statistics.blf.org.uk/asthma (Accessed 09 February 2022).

hospital admissions occur these are usually due to pneumonia rather than acute bronchitis itself.

The CRF comes from the WHO HRAPIE project recommendations in 2013 (WHO,2013[72]) based on pooled data from several panel studies in children (Hoek et al 2012). Data on baseline rates of acute bronchitis in children is not collected routinely so we used the mean daily prevalence from the Pollution and the Young (PATY) study as recommended in WHO (2013) (Table 17).

Table 17 Calculation inputs for acute bronchitis in children

Population at risk, children **age 6-12**	Baseline daily prevalence of acute bronchitis in children	Concentration-response function
UK Total age 6-12 for each year 2018 – 2134 Derived from single year of age population data generated by the life table calculations	Mean prevalence from the Pollution and the Young (PATY) study: 18.6% (range 6–41%)	OR 1.08 (0.98–1.19) per 10 µg m^{-3} **PM_{10}** (Hoek et al 2012, PATY study) recommended by WHO, 2013[73]

The calculation method is that used for studies based on logistic regression – see section on chronic bronchitis/phlegm.

Health outcomes considered but not included

Asthma incidence

Asthma incidence was not included because symptoms in asthmatics and asthma admissions (as part of respiratory hospital admissions) are quantified already. The fact that there is an association with asthma incidence (Gehring et al 2015; Liu et al 2021) but not prevalence (Molter et al 2015; Fuertes et al 2020) from large studies including several cohorts across Europe suggests that air pollution may be affecting a subset of asthma that is more reversible than the classic allergic asthma that starts in childhood and lasts a lifetime. If it is more reversible then it would be less appropriate to quantify asthma incidence (with extra willingness to pay to avoid having a chronic disease) separately from the symptoms and hospital admissions.

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[72]WHO (2013), Health risks of air pollution in Europe-HRAPIE project, WHO Regional office for Europe. Available at: http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/publications/2013/health-risks-of-air-pollution-in-europe-hrapie-project.-recommendations-for-concentrationresponse-functions-for-costbenefit-analysis-of-particulate-matter,-ozone-and-nitrogen-dioxide (Accessed 09 February 2022).

[73]WHO (2013), Health risks of air pollution in Europe-HRAPIE project, WHO Regional office for Europe. Available at: http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/publications/2013/health-risks-of-air-pollution-in-europe-hrapie-project.-recommendations-for-concentrationresponse-functions-for-costbenefit-analysis-of-particulate-matter,-ozone-and-nitrogen-dioxide (Accessed 09 February 2022).

Lung cancer incidence

While there is evidence of associations between air pollution and lung cancer incidence (Hamra et al 2015), lung cancer is unfortunately almost always fatal within a few years. From the perspective of the number of cases and life cut short, this outcome is therefore already counted within the all-cause mortality calculations. This may omit some economic aspects such as health care costs but lung cancer is quite rare relative to other causes of mortality and health care costs are usually small compared with willingness to pay for mortality.

Stroke incidence

There is some evidence of an association between air pollution and stroke (Stafoggia et al, 2014) (not statistically significant in the overall analysis but significant in some sub-groups), COMEAP (2021a)[74] reported a meta-analysis for air pollution and stroke incidence that was just not statistically significant but was supported by mechanistic evidence. An association is plausible given the wider evidence on air pollution and cardiovascular disease and the fact that cardiovascular disease is a risk factor for stroke. However, cardiovascular disease being a risk factor for stroke leads to the possibility of double counting in monetary valuation of the effects. It might be possible to disentangle this, but this would have required more time than was available. This is important for further work because the health care costs for stroke are significant.

Diabetes incidence

The literature on air pollution and diabetes is relatively new. The evidence has not been discussed formally by COMEAP. It is also complicated by the fact that diabetes is a risk factor for cardiovascular disease, leading to potential for double counting. We did not include this for this short project but recommend further detailed consideration in future work.

Dementia incidence

There is a developing literature on air pollution and dementia incidence but it is somewhat contradictory with respect to the evidence on each pollutant separately (COMEAP, 2017). A COMEAP report on dementia is due to be published soon and this health outcome could be considered further after that. Again, there are potential issues with overlap with cardiovascular disease, which is a risk factor for dementia.

Comparison with health outcomes used in Defra damage costs

The Defra damage costs guidance[75][76] includes many of the same outcomes as described here. There are some differences with respect to morbidity outcomes. Defra include asthma, lung cancer, stroke and diabetes, which we did not, for the reasons given above.

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[74] https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/1002468/COMEAP_Env_Bill_PM2.5_targets_health_evidence_questions_responses.pdf (Accessed 09 February 2022).

[75] https://www.gov.uk/government/publications/assess-the-impact-of-air-quality/air-quality-appraisal-damage-cost-guidance

[76] https://uk-air.defra.gov.uk/assets/documents/reports/cat09/2007031424_Damage_cost_update_2020_FINAL.pdf

For coronary heart disease, Defra uses HR 1.19 (1.01; 1.42) per 5 µg m^{-3}. This was converted from 1.41 (1.00 - 2.01) per 10 µg m^{-3} from the UK Public Health Forum report (Public Health England, 2018[77]), which was in turn based on Cesaroni et al 2014. The COMEAP advice we used is based on a meta-analysis that includes Cesaroni et al 2014 but also other studies.

We also took a different approach for chronic bronchitis in relation to valuation, although the CRF was the same, Defra followed the COMEAP advice to include the full disease of chronic bronchitis in sensitivity analysis, while we included it in core analysis but as milder symptoms.

[77]https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/836720/Estimation_of_costs_to_the_NHS_and_social_care_due_to_the_health_impacts_of_air_pollution.pdf (Accessed 09 February 2022).

8. Methods to value health and economic impacts

Air pollution is the largest environmental health risk to the UK (Defra, 2019[78]). Air pollution both shortens lives and contributes to chronic illness. Studies have estimated that air pollution is responsible for 19% of all cardiovascular deaths and 29% of all lung cancer deaths (CBI, 2020). This imposes costs both to individuals that experience these conditions, and to societies and economies as a whole. The CBI[79] (2020) estimates that reducing air pollution in the UK would result in people living and working longer, and contributing an additional £1 billion to the economy in the first year, and larger gains in later years if early retirement is avoided[80]. Figure 9 sets out the channels through which air pollution affects health and economic outcomes, and Table 18 focus on the channels covered in this study.

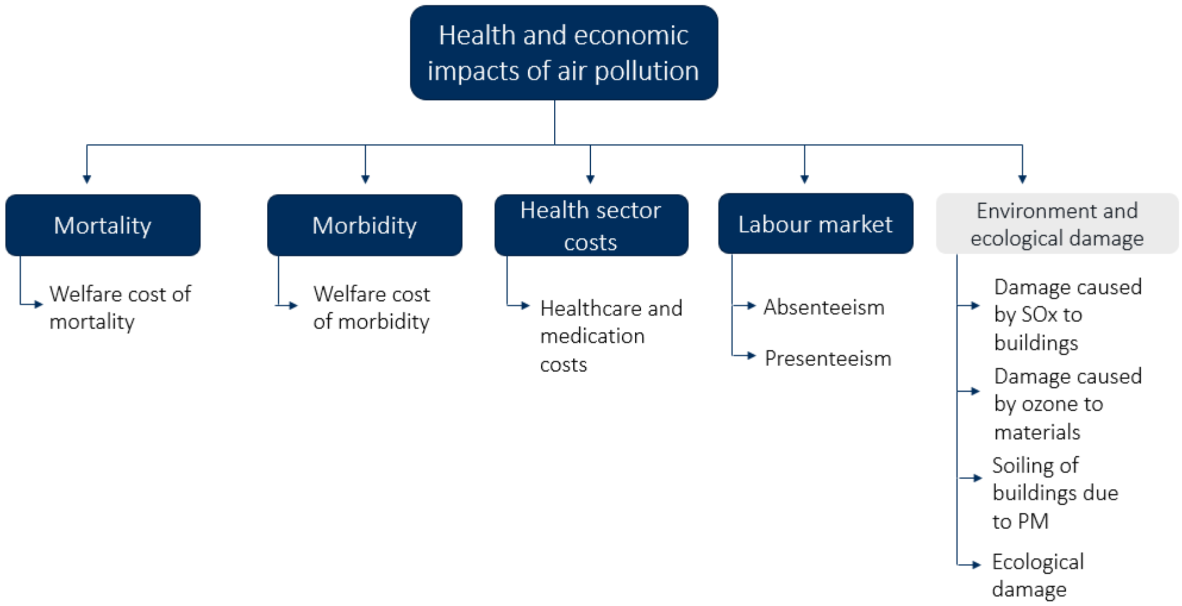

Figure 9 Channels through which air pollution affects health and economic outcomes

These channels were selected based on Defra guidance for assessing the impact of air pollution (Defra 2021[81]). The air pollution modelled in this study did not include ozone and SOx since our focus was on the forecasts of $PM_{2.5}$ in 2030, although we also used PM_{10} and NO_2 in the health analysis.

This study focuses on four channels to estimate the impact of air pollution on economic and health outcomes, summarised in Table 18.

[78]Clean Air Strategy

[79]https://www.cbi.org.uk/media/5539/2020-09-cbi-economics-caf-report.pdf (Accessed 09 February 2022).

[80]The method adopted for this study is different from that adopted by the CBI, which makes direct comparisons difficult.

[81]https://www.gov.uk/government/publications/assess-the-impact-of-air-quality/air-quality-appraisal-impact-pathways-approach (Accessed 09 February 2022).

Table 18 Channels of economic impact and its measurement

Channel	Indicator	Measurement
Mortality		
Welfare cost of mortality	Life years lost Post neonatal mortality (1- 12 months)	Willingness to pay studies that value life years lost Willingness to pay studies on post neonatal mortality
Morbidity		
Welfare cost of morbidity	Time spent in hospitals for specific illnesses Disease incidence, prevalence or symptom days	Both indicators are measured using willingness to pay studies that ask people to place a monetary value on time spent in hospitals, and the impact of certain diseases.
Health sector costs		
Healthcare system costs	Prevalence of disease in a particular year	Costs to the healthcare system from ill health (includes the cost of primary care, secondary care, palliative care and medication costs) based on a study funded by Public Health England and UK Medical Research Council (Pimpin et al., 2018).
Labour market costs		
Labour market costs	Symptom days	The economic cost of absenteeism and presenteeism is based on estimating lost economic output from workplace absences and lower productivity due to ill health.

8.1. Welfare gains from reducing premature mortality

The economic impacts of premature mortality are assessed based on the value society is willing to pay to avoid premature death. Total premature mortality is expressed as life-years lost across the population as a result of premature deaths. Premature deaths are captured by assessing the timing of death relative to that in other scenarios, expressed as a change in life years. The calculations and analysis underpinning the assessment are explained in more detail in the Health Impact Assessment Methods (Section 7).

The economic impacts of air pollution related mortality can be quantified by assigning a monetary value to the life years lost. These values are based on revealed and stated preference methods that elicit estimates of what individuals are willing to pay or accept for a certain health outcome (UK Department of Health, 2010[82]).

- **Revealed preference** methods involve the observation of market situations in which people trade wealth or income against the risk of death or injury. This method typically uses multiple regression analysis to isolate the risk effect from the other factors that vary across individuals, to estimate the market value of the risk of death.

- **Stated preference or contingent valuation** methods ask individuals their willingness to pay for or accept changes in the risk of death. This method involves surveys to establish people's preferences by presenting hypothetical market situations where people may 'purchase' a reduction in the probability of an accident or 'sell' an increase in that probability.

Using these monetary values and the modelling outputs from Imperial College, we calculate the welfare loss due to premature mortality. Our analysis makes an additional adjustment for post neonatal mortality (for infants between 1 -12 months) associated with air pollution. This is because neonatal mortality is estimated based on incidence rather than life years lost.

Figure 10 Estimating the impact of welfare loss due to premature mortality

This analysis uses monetary values recommended by the UK Government to estimate the economic value of premature mortality. We use two values recommended by the Government to develop a range of estimates for the value of reducing premature mortality.

[82]https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/216003/dh_120108.pdf (Accessed 09 February 2022).

- **Defra guidance:** Defra values a life year lost from air pollution at £43,626[83]. The valuation is based on a willingness to pay study conducted in 2004 (Chilton, 2004).
- **HM Treasury guidance:** The HMT Green Book – the government's guide to cost-benefit analysis – advises using a value of £60,000[84] for when assessing life years lost or gained in economic impact assessments.

The differences in values are likely due to different methodological approaches to assessing willingness to pay. There are also likely to be differences in the application of weights to individual ages.

Monetary valuation of post neonatal life lost is based on Walton et al 2015 rebased to a 2018 price base (£ 3,294,542 in 2018 prices).

8.2. Welfare gains from reducing morbidity

Morbidity is the level of illness in the general population and results in both a reduction in quality of life and total years lived[85]. The benefits of reducing morbidity are measured using willingness to pay studies that estimate the welfare loss to individuals.

The economic analysis uses multiple outputs from the health impact analysis, depending on the disease assessed. The choice of diseases and the indicators used to estimate health impact follow guidance provided by the Committee on the Medical Effects of Air Pollutants (COMEAP, www.comeap.org.uk), and the World Health Organisation (WHO, 2013). These are summarised in Table 19.

- **Incidence** measures the change in new cases
- **Prevalence** measures the total number of people with the disease, including both new and existing cases
- **Symptom days** measures the number of days an individual suffers symptoms associated with a particular disease

[83] Air quality appraisal: impact pathways approach 2017 valuation for chronic mortality (https://www.gov.uk/government/publications/assess-the-impact-of-air-quality/air-quality-appraisal-impact-pathways-approach) rebased for 2018 (Accessed 09 February 2022).

[84] HMT 2020 A1.51 reported as value for a statistical life year (SLY) available (https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/938046/The_Green_Book_2020.pdf) (Accessed 09 February 2022).

[85] https://www.gov.uk/government/publications/assess-the-impact-of-air-quality/air-quality-appraisal-impact-pathways-approach (Accessed 09 February 2022).

Table 19 Choice of disease and indicators used

Indicator	Disease
Incidence	Coronary heart disease, acute bronchitis in children
Prevalence/symptom days	Chronic bronchitis measured through chronic phlegm (prevalance) Asthmatic symptoms, bronchitic symptoms in asthmatic children (symptom days)
Hospital admissions	Cardiovascular diseases, respiratory diseases

The indicators selected for disease type are based on COMEAP guidance and a review of the health literature. This provides the evidence base to quantify the impact of air pollution.

The health impacts are monetised based on:

- Welfare loss to individuals due to ill health
- Welfare loss to individuals due to hospitalisations

The welfare loss to individuals is typically separated into the loss associated with ill health and the loss associated with hospital admissions because they capture different impacts on individuals. The welfare loss due to ill health captures the day to day suffering of people living with a disease for a whole year. The welfare loss due to hospitalisations represent the disutility associated with a single hospital admission event[86]. For this reason, there is unlikely to be a significant overlap between the welfare effects associated with having symptoms of the disease and those associated with hospital admissions.

Welfare loss due to ill health

Welfare loss is based on the principle of a decline in utility for individuals that suffer from a disease. Figure 11 sets out the broad approach adopted.

Figure 11 Estimating the impact of welfare loss due to ill health

[86] https://www.gov.uk/government/publications/assess-the-impact-of-air-quality/air-quality-appraisal-impact-pathways-approach (Accessed 09 February 2022).

For specific diseases we set out the approach and our assumptions by disease type in Table 20, *Table 21*, *Table 22*, *Table 23* and Table 24:

Table 20 Chronic Bronchitis (chronic phlegm)

		Value	Source/Method
A	Prevalence of disease by year	Modelled annually	Imperial modelling
B	Duration of disease	More than 3 months – assumed	See below
C	Monetary value of symptom day	£54 per day	Walton (2015) rebased to 2018[87]
D	Monetary valuation	Calculation	D= A*B*C

The definition of chronic phlegm used in Cai et al (2014) (see health impact assessment methods section 7) is symptoms of chronic phlegm for at least 3 months of the year for 2 years in a row. We used this information to derive the monetary valuation on the basis of 90 symptom days. This is for 1 year not 2, because the valuation was applied each year.

Table 21 Coronary Heart Disease

		Value	Source/Method
A	Incidence of disease	Modelled annually	Imperial modelling
B	Welfare loss as a result of incidence	£208,963	Incidence value from reverse QALY calculation based on Defra approach[88]
C	Monetary valuation	Calculation	C= A*B

[87] https://www.london.gov.uk/sites/default/files/hiainlondon_kingsreport_14072015_final.pdf (Accessed 09 February 2022).

[88] Based on Defra QALY and expected duration of disease https://uk-air.defra.gov.uk/assets/documents/reports/cat09/2007031424_Damage_cost_update_2020_FINAL.pdf (Accessed 09 February 2022).

Table 22 Acute Bronchitis

		Value	Source/Method
A	Incidence of acute bronchitis	Modelled annually	Imperial modelling
B	Welfare loss as a result of incidence	£3,419	Acute bronchitis valuation children Walton (2015)[89] rebased to 2018
C	Monetary valuation	Calculation	C= A*B

As explained in the health impact assessment methods section acute bronchitis can last for up to 3 weeks, but is not usually serious enough to require admission to hospital. The economic valuation takes this into account.

Table 23 Asthmatic symptom days

		Value	Source/Method
A	Symptoms of asthma	Modelled annually (symptom days)	Imperial modelling
B	Welfare loss as a result of incidence	£82	LRS valuation (children) from Walton (2015)[90] rebased to 2018
C	Monetary valuation	Calculation	C= A*B

[89] https://www.london.gov.uk/sites/default/files/hiainlondon_kingsreport_14072015_final.pdf (Accessed 09 February 2022).

[90] https://www.london.gov.uk/sites/default/files/hiainlondon_kingsreport_14072015_final.pdf (Accessed 09 February 2022).

Table 24 Bronchitic symptoms in asthmatic children

		Value	Source/Method
A	Symptoms of bronchitis	Modelled annually (symptom days)	Imperial modelling
B	Symptoms of bronchitis	£114	Assumed acute bronchitis valuation over a 30 day period and applied to asthmatic children experiencing Bronchitic symptoms
C	Monetary valuation	Calculation	C= A*B

Welfare loss due to hospitalisation

The economic impact of hospital admissions focuses on the welfare gains from reduced respiratory and cardiovascular diseases because the causal pathway between air pollution and incidence is well established in the academic literature. Air pollution is also associated with other diseases including lung cancer, however the research is less well established and therefore these diseases are not included in the economic analysis. Figure 12 sets out the approach adopted.

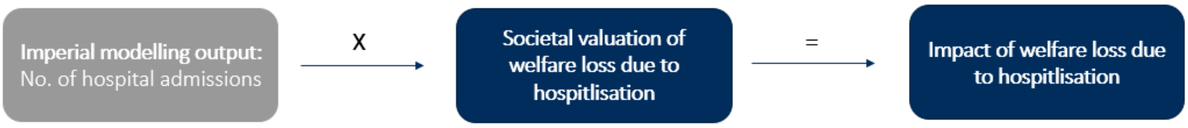

Figure 12 Estimating the impact of welfare loss due to hospitalisation

The analysis for welfare loss due to hospitalisation is based on current Defra guidance that uses updated figures from Chilton (2004). The values from the Chilton (2004) study are based on a survey that asked participants about their willingness to pay to avoid hospitalisation. The updated values for 2018 are:

- £8,460 per respiratory admission
- £8,638 per cardiovascular admission

8.3. Estimating health sector costs

This analysis estimates potential reduced costs for the healthcare system from reduced incidence of disease. This study uses Public Health England and UK Medical Research Council commissioned analysis to estimate of the costs of primary, secondary, and palliative cases for different diseases to assess the potential cost savings as a result of improved health outcomes.

This study does not quantify the second order impacts of air pollution related health expenditure. For example, individuals that are able to live longer because of reductions in air pollution might impose additional costs on the health service in their old age. Depending on the illness, in some cases these longer term costs could be higher than savings realised in the short-term.

Figure 13 Estimating the costs to the healthcare system

To estimate the cost to the health sector we have focused on two diseases (coronary heart disease and chronic bronchitis) where these costs can be quantified by converting our assessed indicators to the cost data provided by the Public Health England commissioned study. The methodology for these calculations are stated below.

Coronary Heart Disease

		Value	Source/Method
A	Incidence of disease	Modelled annually	Imperial modelling
B	Average duration of a disease	9.5 years	Based on Defra 2019[91]
C	Cost of treating a case in a year	£1,200	Based on data gathered as part of PHE commissioned work (Pimpin et al., 2018)
D	Monetary valuation	Calculation	D= A*B*C

[91]https://uk-air.defra.gov.uk/assets/documents/reports/cat09/2007031424_Damage_cost_update_2020_FINAL.pdf (Accessed 09 February 2022).

Chronic Bronchitis (chronic phlegm)

		Value	Source/Method
A	Prevalence of disease in a year	Modelled annually	Imperial modelling
B	Cost of treating a case in a year	£136	Based on data gathered as part of PHE commissioned work (Pimpin et al., 2018) for Asthma treatment
C	Monetary valuation	Calculation	C= A*B

8.4. Labour market impacts

Labour market impacts measures the impact of ill health on the economy through two channels:

- **The economic cost of absenteeism:** Absenteeism is when people in the working population take time off work because of ill health. People may also need to take time off work to take care of any dependents. This could result in a loss of economic output and/or loss of income to the individual[92].
- **The economic cost of presenteeism:** Presenteeism occurs when people attend work when ill, which can reduce their productivity.

This analysis was unable to consider the impact of ill health on early retirement. Therefore, our estimates are conservative, and the true impacts of air pollution on workers and businesses could be much larger.

As part of the air pollution modelling, Imperial have estimated the number of symptom days for diseases caused by air pollution. For each day that an individual has a symptom associated with an air pollution related disease, they either take time off work (absenteeism) or attend work but perform at a lower level of productivity (presenteeism).

Our analysis assumes that on days that patients exhibit symptoms they are:

- Present at work for 75% of these symptom days, but not performing at their expected level of capacity. Our assumption is based on a CIPD 2020 report[93] which reports

[92]This analysis assumes one day of absence from work results in a day of lost economic output. However, in practice, the relationship is likely to vary across firms and sectors. In some cases, work place absence could lead to an individual's work being picked up by other team members in a way that reduces the overall economic loss. Evidence to support this point is based on research from a parental communication and advocacy forum: https://www.mumsnet.com/news/parents-use-annual-leave-or-take-unpaid-leave-to-care-for-sick-children (Accessed 09 February 2022).

[93]https://www.cipd.co.uk/Images/health-wellbeing-work-report-2021_tcm18-93541.pdf (Accessed 09 February 2022).

observations of “present” workers who are working despite being ill. We assume that workers experiencing symptoms while at work are 20% less productive than they would otherwise be, based on a literature review by Defra[94].

- On the remaining 25% of symptom days workers are assumed to be absent.

We use two variables to estimate the economic impact of absenteeism and presenteeism.

- Adjust for unemployment: Some individuals that suffer from air pollution related symptoms could be out of the work. To reflect this, we make an adjustment for unemployment using data from the ONS 2018[95].
- The economic value of lost work is based on hourly labour costs in 2018 from ONS (£19.8 per hour per day[96]), and an assumption that the working day is 5.4 hours. This results in a cost of £107.5 per day absent, which is applied to both the working population, and working parents of sick children.

We make the following additional assumptions for child symptom days:

- 59.7% of children have economically active parents that would result in a working day lost for one parent. This is based on the number of children in working households (ONS, 2018[97]) i.e. where all individuals over the age of 16 are employed.

8.5. Inflation, rebasing and discounting

Cost benefit analysis and economic appraisals often source data from different years e.g. much of the evidence base, particularly for mortality valuation, is drawn from a 2004 study (Chilton, 2004). To compare these values with today’s value (or to 2018 as required by this analysis) certain adjustments need to be made.

- **Inflation and deflation:** The HMT Green Book advises using real prices for cost benefit analysis. This requires making an adjustment to remove the effects of inflation from the prices used in the analysis to ensure that the prices used capture real changes. This allows values to be compared consistently over time.

In line with HM Treasury guidance, we rebase historic and current prices using the UK GDP deflator. This allow us to convert all prices into a consistent base year (2018 in this case).

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[94] https://uk-air.defra.gov.uk/assets/documents/reports/cat19/1511251135_140610_Valuing_the_impacts_of_air_quality_on_productivity_Final_Report_3_0.pdf (Accessed 09 February 2022).

[95] https://www.ons.gov.uk/employmentandlabourmarket/peopleinwork/employmentandemployeetypes/bulletins/uklabourmarket/november2018 (Accessed 09 February 2022).

[96] Q4 2018 https://www.ons.gov.uk/employmentandlabourmarket/peopleinwork/earningsandworkinghours/bulletins/indexoflabourcostsperhourilch/julytoseptember2020#the-value-of-labour-costs (Accessed 09 February 2022).

[97] https://www.ons.gov.uk/employmentandlabourmarket/peopleinwork/employmentandemployeetypes/bulletins/workingandworklesshouseholds/apriltojune2018 (Accessed 09 February 2022).

The deflator measures the price level of all goods and services of the base year relative to other years and measures the total goods and services produced in the economy (thereby measuring for real growth). Real growth is used as a proxy for the wealth effect assuming a constant relationship between societal wealth and willingness to pay for health outcomes. In reality this relationship may not be linear and societal valuation of health outcomes might increase after certain wealth thresholds are reached.

Discounting

Discounting allows costs and benefits with different time spans to be compared on a common 'present value' basis. The discount rate captures the idea that society prefers consumption today rather than in the future, therefore short-term benefits are valued more than long-term benefits. The UK Government uses a Social Time Preference Rate (STPR) of 3.5% for government appraisals. This is broken down into two key components:

- **Time preference:** the rate at which consumption and public spending are discounted over time, assuming no change in per capita consumption. This captures the preference for value now rather than later.

- **Wealth effect:** This reflects expected growth in per capita consumption over time, where future consumption is expected to be higher than current consumption, and is expected to have lower utility.

The use of discounting for government appraisals differs from that used in the private sector. In the public sector, the overall size of spending and the allocation of budgets are taken on a top-down basis. When appraising the costs and benefits of individual projects, the government does not consider the cost of raising funds (e.g. through general taxation or issuing debt). This is different from the private sector, where project appraisals consider the cost of raising capital and compensation of risk.

In line with HMT guidance on health assessments, we have used a discount rate of 1.5% for this analysis, diminishing to 1.29% after 30 years. After 60 years an adjustment is made to reduce the discount factor 1.07% and it remains at this level for the duration of our analysis. Consequently, an economic value in 2134 (the last year for this analysis) is worth about 23% of the economic value in 2018. Consistent with HMT guidance, this analysis excludes the wealth effect component of the STPR. This is because the principle of diminishing marginal returns does not apply to life years gained, as the utility associated with life years gained does not decline as incomes rise. It is for the same reason that we do not include an up-lift factor that increases values over time in line with per capita income growth.

9. $PM_{2.5}$ forecasts in the UK between 2018 and 2030

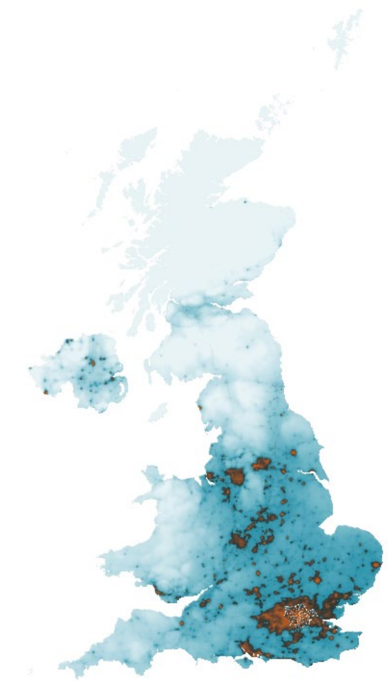
The UK model forecasts of $PM_{2.5}$ in 2018 (Figure 14, top left panel) show that there are large areas that already comply with the WHO-10 interim target: northern and western England, Scotland, Wales and Northern Ireland. However, in major cities and in a line from roughly York and Manchester, heading south and east there are a number of locations above WHO-10. Some are associated with major cities and close to major roads, but others are primarily associated with the use of biomass for combustion within industry, and seen as 'spots' of $PM_{2.5}$ throughout England and Northern Ireland. In contrast, the 2030 forecasts show widespread compliance with the WHO-10 interim target value throughout the UK, and that exposure to $PM_{2.5}$ above WHO-10 is limited to locations close to roads in major cities and to some of the larger sources of biomass burning in industrial processes (*Figure* 14, top right-hand panel). Overall, 6.4% of the area of the UK still had concentration exceeding the WHO-10 interim target value in 2018; this figure is predicted to reduce to 0.2% in 2030 scenario UK2030+LS1. In the forecasts of 2030, we have adopted a similar approach to that of compliance with EU limit values such as for NO_2, i.e., we have removed the concentrations within each roadway, since this is not considered to be relevant to human exposure. The predictions of Manchester and Glasgow in Figure 14 show the network of roads (in white), representing the centres of roads that have been removed.

In the bottom panels of Figure 14 are three examples of our 2030 predictions, in the West Midlands, in Manchester and in Glasgow. In Manchester, in 2030 and within the M60 ring road, the urban background concentrations are typically ~8.5 $\mu g\ m^{-3}$, with a range of ~8.2 to 8.9 $\mu g\ m^{-3}$. The highest $PM_{2.5}$ concentrations are close to the busiest major roads, even after removal of concentrations within the roads themselves, with a risk of exceeding WHO-10 at these locations. Whilst the model has a detailed assessment of major roads in the city, there is in reality a higher density of roads in Manchester than has been modelled, and this may provide other high-exposure locations.

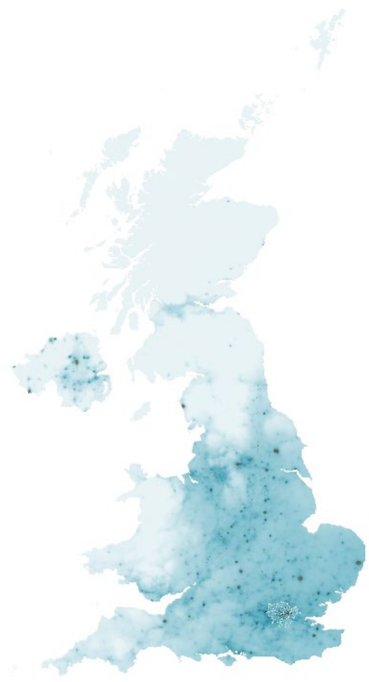
In the Birmingham and the West Mildands region, in 2030, which includes Wolverhampton, Walsall, Dudley, Sandwell, Birmingham, Solihull and Coventry, the urban background concentrations are typically ~8 $\mu g\ m^{-3}$, with a range of ~7.2 to 8.6 $\mu g\ m^{-3}$. Once again, the highest $PM_{2.5}$ concentrations are close to the busiest major roads, even after removal of concentrations within the roads themselves, with a risk of exceeding WHO-10 at these locations. Whilst the model has a detailed assessment of major roads in the region, there is also a higher density of roads than has been modelled, and this may provide other high-exposure locations.

For many UK cities there is a similar picture to the one given for the West Midlands and Manchester, except that in the case of Glasgow the urban background concentrations are lower, being typically around 5 to 6 $\mu g\ m^{-3}$. Once again though the highest PM concentrations close to the busiest roads remain in excess of the WHO-10 target value, whilst the roadside concentrations on the smaller roads are often below it.

UK $PM_{2.5}$ in 2018



UK $PM_{2.5}$ in 2030



Legend

Manchester in 2030

Glasgow in 2030

Figure 14 CMAQ-urban UK and city model results for $PM_{2.5}$ in 2018 and 2030. The results show considerable improvement in the concentration of $PM_{2.5}$ between the 2018 base year and 2030 assuming DEFRA's business as usual scenario, combined with vehicle emissions based upon the CCC's Balanced Net Zero Pathway.

The change in $PM_{2.5}$ concentrations across the UK between 2018 and 2030 is brought about through reductions in both local primary $PM_{2.5}$ (emitted locally) and secondary $PM_{2.5}$, which is derived from precursor emissions released 10s to 100s km away. Of the secondary $PM_{2.5}$ changes between 2018 and 2030, the most important is Secondary Inorganic Aerosols (SIA – nitrate, sulphate and ammonium) whose median concentration across the UK reduced by -0.86 µg m^{-3}. Other important changes to $PM_{2.5}$ include "other" (median change -0.27 µg m^{-3}), organic aerosol, which can be both primary and secondary $PM_{2.5}$ (median change -0.15 µg m^{-3}), and elemental carbon (median change -0.08 µg m^{-3}). In this case, "other" represents unclassified $PM_{2.5}$ components including metals and mineral dust. There is also a zero change in sea salt, which is because the same meteorology was used in both 2018 and

2030. Whilst these results suggest the importance of controlling NO_x, NH_3, and SO_2 (i.e., SIA precursors) emissions to meet WHO guidelines by 2030 they also hide a wide range of possible changes in $PM_{2.5}$ from rural/remote locations in Scotland where changes are small (~0.2/3 µg m^{-3}) to much larger changes in city centres (>2 µg m^{-3}). Since the largest changes to $PM_{2.5}$ also reflect the largest populations, this in turn influences the population-weighted average concentrations presented below.

9.1. Population-weighted average $PM_{2.5}$ concentrations (PWAC)

The population-weighted average concentration has been calculated for each country in the UK, the UK as a whole, and for Manchester, Glasgow and London. This is calculated by combining average $PM_{2.5}$ predicted concentrations in each of the 8887 wards in the UK with their associated population, then using population as a weight to compute a weighted average concentration for each local authority area. The local authority PWAC values are then averaged to give values for country and city regions.

PWAC is useful in assessing exposure reduction to $PM_{2.5}$, one of the targets to be set as part of the DEFRA consultation exercise but is not suited to looking at compliance with WHO-10. However, from the $PM_{2.5}$ PWAC data (see Table 25), the exposure to $PM_{2.5}$ between 2018 and 2030 is predicted to reduce by between 0.9 and almost 4 µg m^{-3}, depending on whether you live in Scotland or inner London.

Table 25 Population-weighted average $PM_{2.5}$ concentrations (µg m-3) in countries and cities of the UK

	2018	UK2030+LS1	UK2030+LS2	UK2030+LS3	2018 vs UK2030+LS1	2018 vs UK2030+LS2
England	9.78	7.51	7.45	7.44	-2.27	-2.33
Northern Ireland	7.16	5.75			-1.41	
Scotland	5.07	4.20			-0.87	
Wales	7.43	5.71			-1.72	
UK	9.18	7.08	7.03	7.02	-2.10	-2.15
London	11.27	8.55	7.98	7.90	-2.72	-3.29
Inner London	11.97	9.05	8.27	8.19	-2.92	-3.70
Outer London	10.75	8.19	7.76	7.69	-2.56	-2.99
Greater Manchester	10.72	8.17			-2.55	
Glasgow city	7.32	5.86			-1.46	
West Midlands	9.68	7.38			-2.3	

When weighted by the number of people at risk, 41% of local authorities had $PM_{2.5}$ exposure levels above WHO-10 in 2018. This is predicted to fall to less than 1% by 2030 for scenario UK2030+LS1. Taking this a stage further, the change in each LA's PWAC is illustrated in Figure 15 below. These are ordered from highest to lowest in 2018, with each LA occupying the same x-axis position in 2030. This demonstrates that the impact of the emissions changes between 2018 and 2030 driving important exposure reductions, and also that as you transition from a current high exposure location to low exposure one the change becomes increasingly small. What is also clear from Figure 15 is the impact that LS2 has, with the London local authorities lying below of the 2030_Sc2 line due to the effect of local action. Finally, the 2030 LS2 and LS3 forecasts show that the PWAC for every UK local authority is below the WHO-10 target.

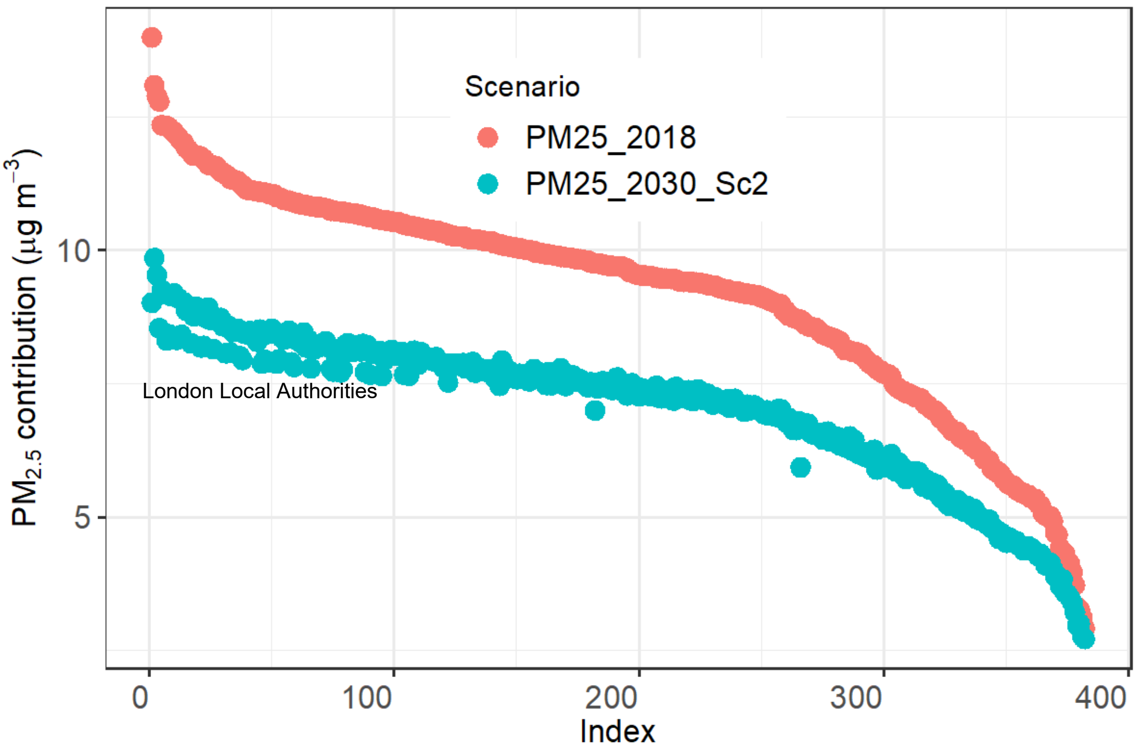


Figure 15 The population weighted average $PM_{2.5}$ concentration (µg m^{-3}) for every local authority in the UK (n=382) – 2018 to 2030

9.2. $PM_{2.5}$ in London in 2018 and 2030

The forecasts of PM in London have been undertaken using the London Toolkit model, allowing us to apportion different London emissions, helping understand their relative importance and undertake three different possible future London scenarios, in order to estimate whether it is possible to comply with the WHO-10 target. The CMAQ-urban and London models were linked in that the former provided the contribution of $PM_{2.5}$ (and PM_{10}, NO_X and NO_2), from sources outside London in both 2018 and for the three 2030 Scenarios.

$PM_{2.5}$ source apportionment by London Borough

To help understand the scenarios that we have tested, we have undertaken a Source Apportionment (SA) analysis of London emissions. SA is helpful in that it splits the total ambient predicted concentrations by emissions source and can be done at a range of scales, from averages across countries and cities down to individual locations. Here we have produced a 2030 SA for each London local authority (LA) (see Figure 16), although it is important to note that for total $PM_{2.5}$ concentrations, a regional contribution of ~7 $\mu g\ m^{-3}$ should be added across all local authorities, demonstrating the important role played by long-range transport of $PM_{2.5}$ from UK and international emissions.

The model results demonstrate that each local authority's ability to control $PM_{2.5}$ varies greatly throughout London, from ~4 $\mu g\ m^{-3}$ in the City to just over 1 $\mu g\ m^{-3}$ in some outer boroughs. The important sources include road transport, despite recent improvements to emissions from this source, cooking, which is important, especially in the centre of the city, domestic wood combustion, construction dust and NRMM, and domestic and commercial gas combustion. Whilst these results, which are based upon the London Atmospheric Emissions Inventory, have helped formulate the three scenarios they also represent some of the most uncertain emissions sources in the UK, and in the case of cooking, a source that is missing from UK emissions altogether.

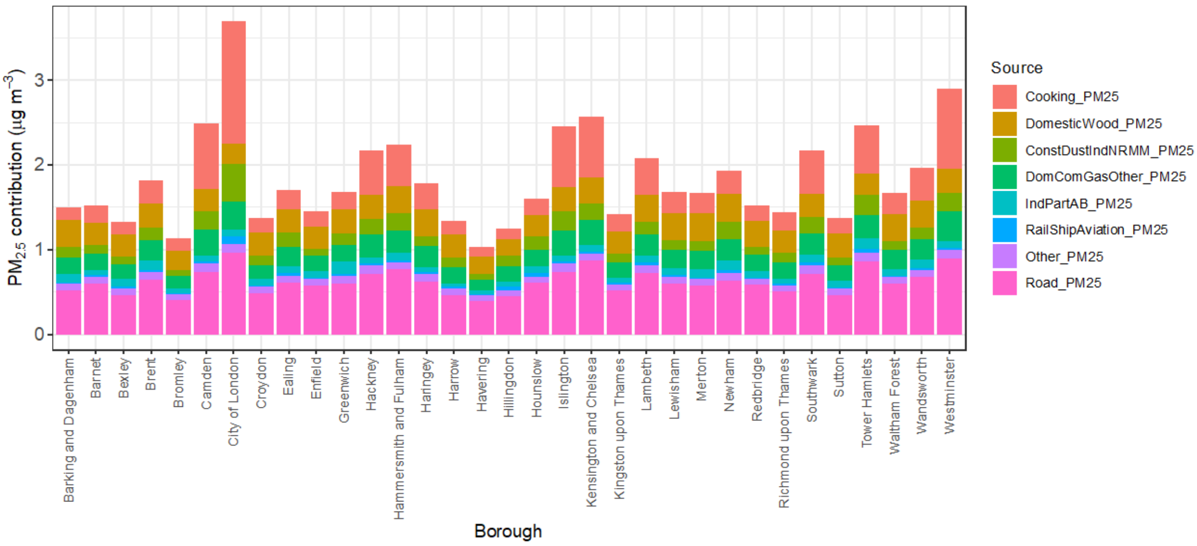


Figure 16 $PM_{2.5}$ Source apportionment in London in 2030 by borough (LS1)

Concentrations of $PM_{2.5}$ for each of the three London Scenarios

As a reminder the three London scenarios are as follows:

- Scenario LS1, which is based upon the London Environment Strategy[98] (LES), includes road traffic changes, such as smaller vehicle km estimates compared with the UK

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[98] https://www.london.gov.uk/sites/default/files/london_environment_strategy_0.pdf (Accessed 09 February 2022).

assumptions, the phasing out of diesel buses and taxis and increased proportions of electric vehicles compared with the UK CCC BNZP. Scenario LS1 also has reductions in emissions from cooking, wood burning, NRMM, domestic and commercial gas/coal and oil combustion, railways/ships (based upon the PLA's Emission Reduction Roadmap[99] and Air Quality Strategy[100] reports) and aviation, agriculture and small-scale waste burning.

- Scenario LS2 extends LS1 to include additional powers required by the Mayor, tackles some non-transport sources and is based upon the Mayor's $PM_{2.5}$ roadmap document[101]. Specifically, LS2 adds further reductions in cooking and domestic wood burning, a ban on burning oil and coal, and reductions in small-scale waste burning emissions.
- Scenario LS3 extends LS2 further by assuming 100% reduction to domestic wood burning.

The concentrations of $PM_{2.5}$ in London for LS1 (see Figure 17), range from ~ 7.5 $\mu g\ m^{-3}$ in the outer areas towards ~9 $\mu g\ m^{-3}$ on the boundary of the congestion charging zone and greater than 10 $\mu g\ m^{-3}$ within the zone itself. Very small points of $PM_{2.5}$ concentrations greater that 10 $\mu g\ m^{-3}$ exist close to major roads beyond the centre of the city, although they are few in number. There are also small areas close to the piers in central London where ships contribute to local exceedences of WHO-10, although these are within the Thames itself. Very large concentrations > 40 $\mu g\ m^{-3}$ exist on the Heathrow site, at the hold point for aircraft taking off, but again this is not related to population exposure.

In LS2, the additional reduction of emissions from wood burning, but in particular from commercial cooking emissions, has a significant bearing on $PM_{2.5}$ concentrations within central London, where reduced concentrations are typically 8.5 to 9 $\mu g\ m^{-3}$ (compared with over 10 $\mu g\ m^{-3}$) and around 7 to 8 $\mu g\ m^{-3}$ for the rest of London.

Many of the benefits of London policy have been realised through scenarios LS1 and LS2 and so the impact of LS3, with its complete ban on domestic wood burning, whilst beneficial is relatively modest. However, even in LS3, there remains a very small number of locations, close to major roads, especially in the centre of London, where the WHO-10 target is exceeded. Overall, 82.6% of London still had very polluted air in 2018; this figure should reduce to 0.61%, 0.021% and 0.017% in 2030 LS1, LS2 and LS3, respectively. For more details on $PM_{2.5}$ concentrations close to roads, see the results of the kerbside concentration analysis in London, below.

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[99] https://server1.pla.co.uk/assets/emissionsroadmapjune2020final.pdf (Accessed 09 February 2022).

[100] https://server1.pla.co.uk/assets/airquality2020v1.pdf (Accessed 09 February 2022).

[101] https://www.london.gov.uk/sites/default/files/pm2.5_in_london_october19.pdf (Accessed 09 February 2022).

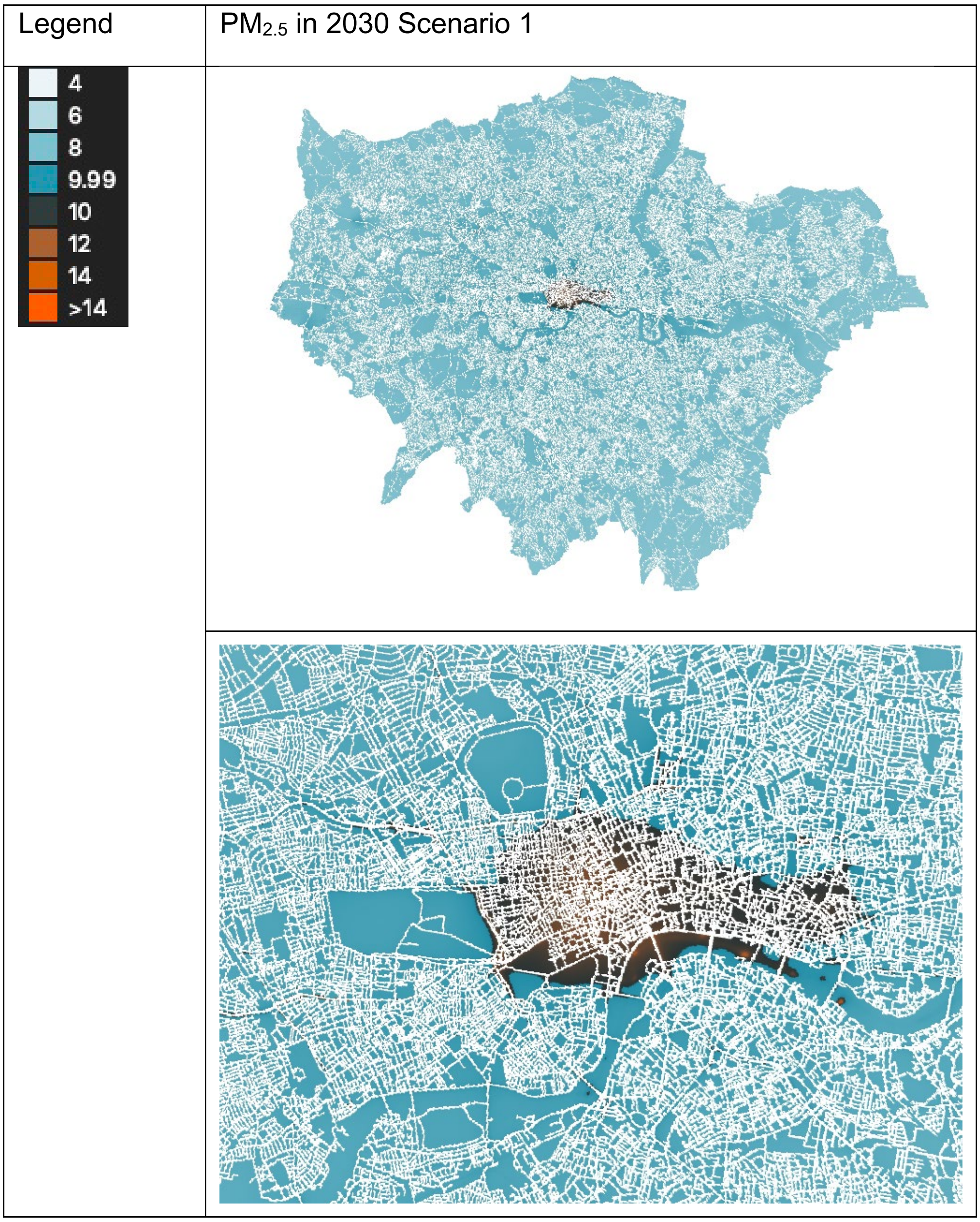


Figure 17 The London model results for $PM_{2.5}$ in 2030 Scenario LS1

Kerbside concentration analysis in London

One of the two DEFRA $PM_{2.5}$ targets relates to compliance with WHO-10 everywhere, including close to roads, where the highest concentrations of $PM_{2.5}$ in cities often occur. In London we have undertaken a detailed analysis of the modelled concentrations within 2m of the kerb of a selection of London's major roads in 2018 (see Figure 18 top panel), as well as for each of the three 2030 London scenarios. The major roads are those used in DEFRA's recent Model Inter-comparison Exercise. The model concentrations have been sampled at 2m from the kerb and averaged along both sides of the road, but without the removal of locations close to junctions. The results are summarised as density plots (see Figure 18 bottom panel). This analysis shows that there are marked improvements in the $PM_{2.5}$ concentrations at kerbside locations between 2018 and 2030, with 2018 showing widespread exceedance of the WHO-10 target, and in 2030, an ever decreasing proportion of the these locations showing $PM_{2.5}$ concentrations above 10 $\mu g\ m^{-3}$. 2030 Scenario LS2 and LS3 are very similar, and show that ~11% of the major roads in London still risk having concentrations over 10 μg m^{-3} within 2m of the kerb.

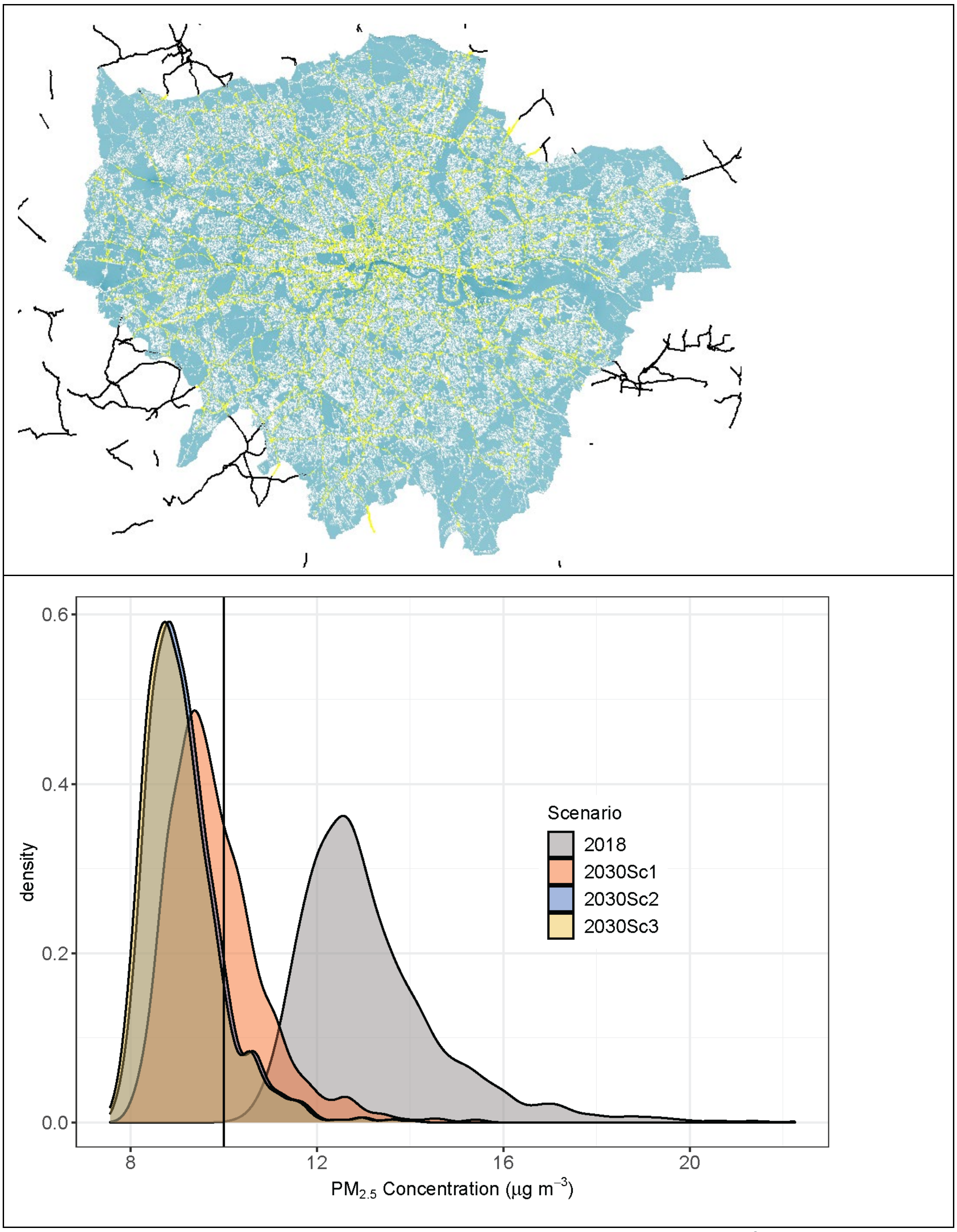


Figure 18 Density plots of kerbside $PM_{2.5}$ concentrations (μg m^{-3}) (bottom panel), for all of the major roads in London (top panel)

9.3. The uncertainty in estimating compliance with WHO-10 using the UK and London models

One area of uncertainty related to $PM_{2.5}$ predictions is particle bound water (PBW). Alongside the predictions of $PM_{2.5}$ components, which are combined to give a total $PM_{2.5}$ concentration, we have added PBW according to the methods of Frank (2006). The method of adding PBW (~0.5 μg m^{-3}) is consistent with both the UK's particle measurements and with UK compliance assessment using those measurements.

However, we also have model uncertainty, and both bias and uncertainty are reflected in the performance against 128 measurement sites in the UK, and at the 26 sites in London. To interpret the model's results we have investigated the distribution of the model's errors and have used that distribution to construct one sided confidence intervals, to determine at what concentration we are 95% confident that our model result is below 10 μg m^{-3}. Note that this is some way below the 10 μg m^{-3} concentration on the map. To create the one sided confidence intervals we have used a boot strapping technique to sample the model and observed concentrations in groups of 128 (UK) and 26 (London) 100,000 times; we have then calculated the 5% quantiles of each of the 100,000 datasets. Finally, these are averaged to give the following overall figures:

- UK – the 5% quantile value is -2.1 μg m^{-3}.
- London – the 5% quantile value is -1.7 μg m^{-3}.

These results mean that in the UK, 5% of the model results are under predicted by 2.1μg m^{-3} or more and so if we predict ~7.9 μg m^{-3} or below, we are 95% confident that we have passed the WHO-10 test. Using the London results means that if we predict 8.3 μg m^{-3} we are 95% confident that we have passed the WHO-10 test. Note that these results only apply to this model run, do not account for the model uncertainty and bias changing in future (up or down) and assume that we get the future emissions correct.

How does this uncertainty estimate affect our interpretation of the UK map?

Figure 19 displays UK 2030 and London 2030 Scenario LS2 predictions, as a way of demonstrating the influence that model uncertainty has on the forecast of compliance with the WHO-10. In the top left panel, no account of model uncertainty has been made and almost complete compliance occurs, except for localised industrial biomass burning emissions, discussed previously, and some exceedences close to major roads.

The top right panel shows the same data, but includes the impact of model uncertainty, giving a very different overall picture. So while the predictions are the same in the right hand panel a cut off of 7.9 μg m^{-3} has been included and shows up clearly as darkly coloured zones. Within these zones the predictions are often some way below 10 μg m^{-3}, but we cannot state with 95% confidence that they will not exceed the WHO-10. However, it is possible to state, with 95% confidence, that less than 5% of the UK will exceed WHO-10 in 2030. Finally, the bottom panel shows the London map with a similar darkly coloured zone (set at 8.3 μg m^{-3}), which also has a risk of exceeding WHO-10, despite our predictions being below that value. For LS1, it is possible to state with 95% confidence that 27.1% of the Greater London Authority area will exceed WHO-10 in 2030; this reduces to 4.3% for LS2.

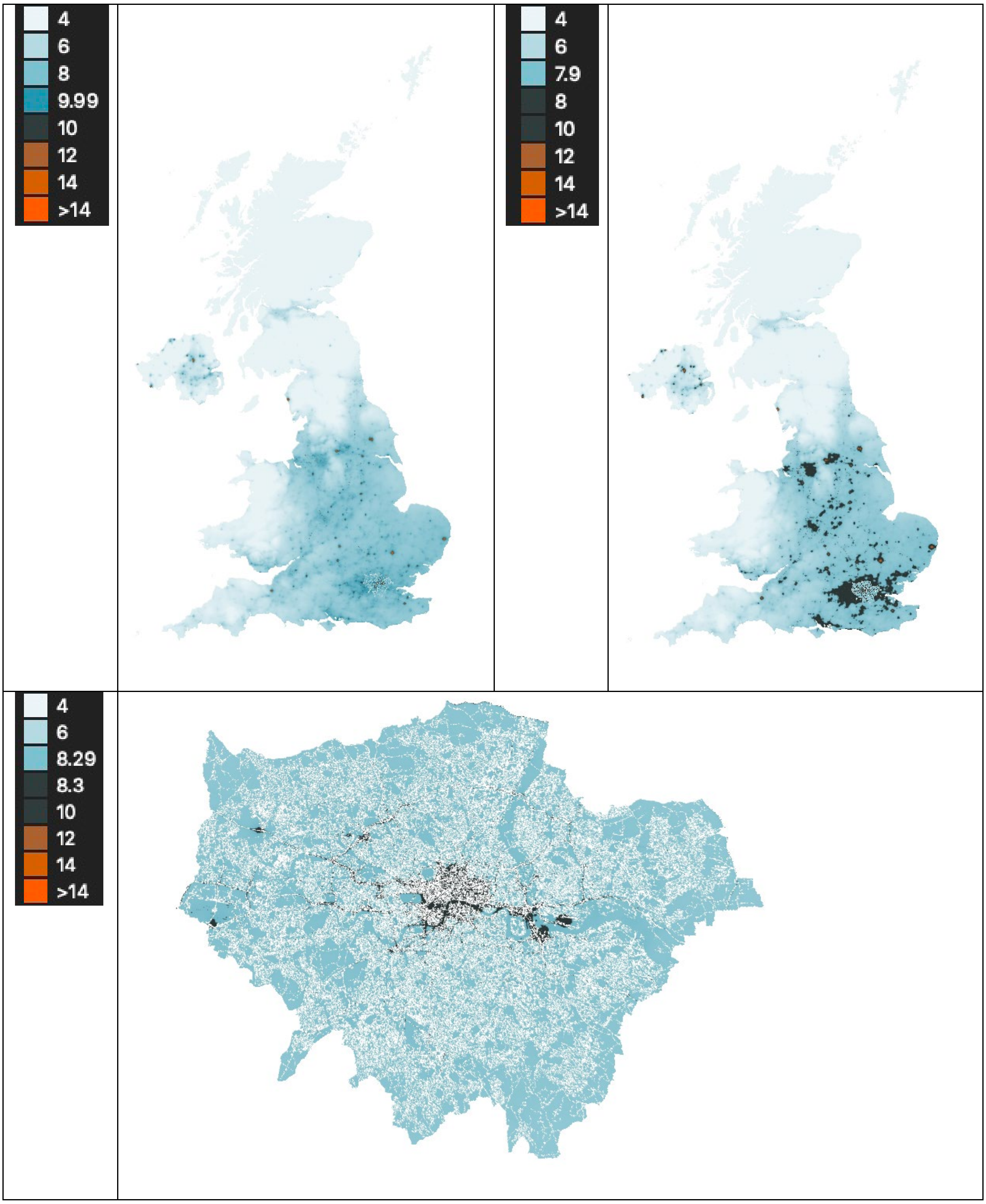


Figure 19 $PM_{2.5}$ predictions in the UK in 2030 with the London predictions in 2030 Scenario LS2. In the top left hand panel no account has been made of model uncertainty whilst the top right and bottom panels include uncertainty

The uncertainty in estimating industrial biomass burning sources

In 2030 there remain 59 industrial sites (down from ~1,500 in 2018), where combustion of wood remains a significant local source of $PM_{2.5}$, and here exceedences of the WHO-10 value occur close to these sites. The industrial sources are typically industrial estates, with multiple sources, cement batching and other small industrial areas, although it is sometimes unclear what the emissions source is. A very small number of these sources, albeit some of the biggest, disappear in the 2019 NAEI (Sellafield and Hinxton) and so should not be considered further in the 2030 estimates. However, since these are important local sources of $PM_{2.5}$ emissions in the NAEI, and without the benefit of any observational evidence close to one of the sources, we have investigated the sensitivity of our predictions to the assumptions used in the model. This was undertaken as part of a separate model run in 2019, and in it we have tested the model's sensitivity to the emissions release height, an important determinant of ground level concentrations. Specifically, we released the PM emissions at ~32m from the ground (between 15 and 50m), rather than at 7.5m (below 15m) as assumed in the 2018 and 2030 runs presented here. This significantly reduced the ground level concentrations from these sources and demonstrates the uncertainty of making these predictions. Whilst the exact details of the release conditions remain unclear, in light of the 2019 sensitivity test, we consider the 2018 estimates to be a 'worst case' prediction, and the 2019 results a better reflection of the true situation regarding these industrial biomass sources. However, ground-based measurements downwind of these sources remains the only certain way of determining the significance of $PM_{2.5}$ related to industrial biomass burning. As a consequence of these tests, and since these industrial sites are outside of major urban areas, and often away from large populations, we have concentrated our analysis on major UK cities.

9.4. Discussion

Our projections show significant reductions in $PM_{2.5}$ concentrations between 2018 and 2030 and with the exception of small zones close to major roads in cities and near to sites of industrial biomass burning, are forecast to be below 10 $\mu g\ m^{-3}$. Across the UK PWAC exposure to $PM_{2.5}$ reduced by ~2 $\mu g\ m^{-3}$ reflecting the greatest changes to $PM_{2.5}$, within large urban populations. In contrast, rural and remote areas of the UK are predicted to have much smaller changes of the order of 0.3 $\mu g\ m^{-3}$.

$PM_{2.5}$ reductions were driven by changes to Secondary Inorganic Aerosols, derived chemically from precursor emissions far away, in combination with changes to local primary $PM_{2.5}$ emissions from sources such as cooking, wood burning and vehicles.

Where our analysis in this report has gone further than that undertaken previously for DEFRA, is the addition of London policies aimed at compliance with the WHO-10. Source apportionment has demonstrated the ability to control $PM_{2.5}$ locally by up to ~4 $\mu g\ m^{-3}$ in the centre of London, as well as demonstrating that some emissions sources such as cooking and domestic wood burning are important, with the former rarely included in future model forecasts. Also, despite recent improvements in emissions performance, road traffic still has an important role to play in exposure to $PM_{2.5}$. Overall the results for London demonstrate the benefits that may be achieved by local action in cities.

We have used the NERC funded CMAQ-urban model, which is uniquely able to predict at a range of relevant scales including close to all major roads in the UK in one go. However, even with this level of detail, it is difficult to provide more than a semi-quantitative assessment of compliance with the WHO-10 target. We have therefore provided population weighted concentrations for every local authority, which in 2030, and as a consequence of the London scenarios, showed that all were forecast to achieve WHO-10. This does not mean that 'everywhere' complies with the standard, and an analysis of kerbside concentrations close to London's major roads revealed that ~11% of roads still risked exceeding 10 $\mu g\ m^{-3}$.

In addition, we have included model uncertainty within the 2030 forecasts we have made, by incorporating one sided confidence intervals created using the distribution of model errors against measurements. Briefly, including a margin of uncertainty in the forecasts results in a very different picture of the risk of exceeding WHO-10, and one where we cannot say that for all areas in the UK we are 95% confident that we can meet WHO-10 in 2030. The results demonstrate the need to reduce model uncertainty, and that PM emissions sources such as vehicle non-exhaust, domestic wood burning and cooking are likely to be important in achieving this.

Finally, it would be hard not to discuss possible future $PM_{2.5}$ concentrations without referring to the natural experiment which is the impact of the COVID lockdown over the last two years. To do this we have compiled a brief analysis of AURN $PM_{2.5}$ data from 2018/19 and 2020, for London, Manchester, Glasgow, Cardiff, Belfast and Birmingham (see Table 26 below). This is meant to reflect what is possible with widespread changes in emissions and is not meant to replicate our 2030 assessment. However, it is clear from these measurements that in 2020, widespread compliance with WHO-10 has occurred. It is also worth mentioning that some of the changes to $PM_{2.5}$ are large, for example in Marylebone Road, whose concentrations have reduced by 7 $\mu g\ m^{-3}$ in 3 years, and for others highlighted, changes of 3-5 $\mu g\ m^{-3}$ in a single year are not uncommon. Whilst some of these changes are as a consequence of a change in $PM_{2.5}$ measurement method, demonstrating the uncertainty in measuring particle concentrations, it is instructive that in 2020 only a few UK sites recorded $PM_{2.5}$ concentrations above WHO-10.

Table 26 2018/19/20 Annual Mean $PM_{2.5}$ concentrations (µg m-3) in UK cities – AURN sites

Site Name	City	2018 $PM_{2.5}$ (µg m^{-3})	2019 $PM_{2.5}$ (µg m^{-3})	2020 $PM_{2.5}$ (µg m^{-3})	Site type	2018 Instrument	2019 Instrument	2020 Instrument
Camden	London	11	11	10	Kerbside	TEOM FDMS	TEOM FDMS	TEOM FDMS
Bexley	London	12	12	9	Urban Background	TEOM FDMS	TEOM FDMS	mixed
Eltham	London	10	11	10	Suburban Background	TEOM FDMS	TEOM FDMS	TEOM FDMS
Honor Oak Park	London	-	10	9	Urban Background	-	Ref.eq	Ref.eq
Marylebone Road	London	16	14	9	Kerbside	TEOM FDMS	TEOM FDMS	TEOM FDMS
N. Kensington	London	9	10	8	Urban Background	Ref.eq	Ref.eq	Ref.eq
Bloomsbury	London	10	11	9	Urban Background	TEOM FDMS	TEOM FDMS	mixed
Harlington	London	10	10	8	Urban Background	Ref.eq	Ref.eq	Ref.eq
Teddington	London	11	12	8	Suburban Background	TEOM FDMS	TEOM FDMS	Ref.eq
Westminster	London	12	12	11	Urban Background	BAM	BAM	BAM
Manchester Piccadilly	Greater Manchester	11	12	8	Urban Background	TEOM FDMS	TEOM FDMS	Ref.eq
Salford Eccles	Greater Manchester	11	9	8	Urban Background	mixed	Ref.eq	Ref.eq
Glasgow High Street	Glasgow	7	6	5	Urban Traffic	mixed	Ref.eq	Ref.eq
Glasgow Townhead	Glasgow	7	7	5	Urban Background	mixed	Ref.eq	Ref.eq
Cardiff Centre	Cardiff	10	12	7	Urban Background	TEOM FDMS	TEOM FDMS	mixed
Belfast Centre	Belfast	10	11	7	Urban Background	TEOM FDMS	mixed	Ref.eq
Birmingham A4540	Birmingham	12	10	8	Roadside	TEOM FDMS	Ref.eq	Ref.eq
Birmingham Acocks Green	Birmingham	9	9	8	Urban Background	TEOM FDMS	mixed	Ref.eq
Birmingham Ladywood	Birmingham	10	10	7	Urban Background	TEOM FDMS	mixed	Ref.eq

Ref.eq – FIDAS monitoring equipment, ‘mixed’ relates to a change in instrument during the year

10. Health results

10.1. UK Mortality impacts

Impacts in the next section are all expressed in terms of life years – the most appropriate metric for the health impact of air pollution concentration changes over time. This used a full life-table approach rather than the short-cut method used for burden. Calculations are first given for $PM_{2.5}$ and NO_2 separately. Because air pollutants are correlated with each other, the air pollutant concentrations in the health studies represent both the pollutants themselves but also other air pollutants closely correlated with them. Health impacts from changes in $PM_{2.5}$ and NO_2 represent the health impacts of changes in the air pollution mixture in slightly different ways that overlap i.e. they should not be added. This is discussed further in this section. $PM_{2.5}$ and NO_2 PWAC data can be found in Table 25 and Table 27, respectively.

Table 27 Population-weighted average NO_2 concentrations (μg m-3) in countries and cities of the UK

	2018	UK2030+LS1	UK2030+LS2	UK2030+LS3	2018 vs UK2030+LS1	2018 vs UK2030+LS2
England	15.14	8.33	8.28		-6.81	-6.86
Northern Ireland	7.10	4.48			-2.62	
Scotland	7.58	4.19			-3.39	
Wales	7.82	4.28			-3.54	
UK	13.85	7.64	7.59		-6.21	-6.26
London	25.78	13.26	12.78		-12.52	-13.00
Inner London	29.63	14.54	13.94		-15.09	-15.69
Outer London	22.95	12.31	11.92		-10.64	-11.03
Greater Manchester	21.91	11.36			-10.55	
Glasgow city	19.69	10.31			-9.38	
West Midlands	15.05	8.45			-6.6	

Table 28 shows the UK results from the life table calculations for anthropogenic $PM_{2.5}$ and NO_2 assuming (i) that the concentration does not reduce from 2018 levels or (ii) that the predicted concentrations changed between 2018 and 2030 (concentrations were modelled at 2018 and 2030 but also interpolated for the intervening years and subsequently maintained at 2030 levels).

If 2018 concentrations of anthropogenic $PM_{2.5}$ remained unchanged for 117 years, around 50.4 million life years would be lost across the UK's population over that period. This improves to around 38.9 million life years lost with the predicted concentration changes between 2018 and 2030 scenario UK2030+LS1 examined here.

The life years lost give a large number because the life years (one person living for one year) are summed over the whole population in the UK or London over 117 years (2018 to 2134). For context, the total life years lived with baseline mortality rates over this period is around 8 billion, so these losses of life years involve about 0.5% of total life years lived.

Another way of representing the health impacts if air pollution concentrations remained unchanged (in 2018) compared with the projected future changes of air pollution up to 2030 (projected from 2018) is provided by the results for NO_2. If 2018 concentrations of NO_2 remained unchanged for 117 years, around 17.9 – 26.1 million life years would be lost across the UK's population over that period. This improves to around 7 – 15 million life years lost with the predicted concentration changes between 2018 and the 2030 scenario UK2030+LS1 examined here.

Summarising these results is not easy. The results should not be added as there is considerable overlap. On the other hand, either result is an underestimate to some extent as it is missing the impacts that are better picked up in the calculations using the other pollutant. COMEAP (2015, 2018a) suggested taking the larger of the two alternatives in the calculation of benefits. We have interpreted this as the larger of the two alternatives ($PM_{2.5}$ or NO_2) on the basis of the central estimate and within the with or without cut-off category. All the relevant data are in the tables to enable creation of summaries in a different form.

In summary, for the projected future changes in air pollution concentrations up to 2030, scenario UK2030+LS1 (projected from 2018) would be around 38.9 million life years lost for the UK population over 117 years.

Table 28 Total life years lost across the UK population for anthropogenic $PM_{2.5}$ and NO_2 for the 2018 baseline and the 2030 scenario UK2030+LS1, scenario UK2030+LS2 and scenario UK2030+LS3

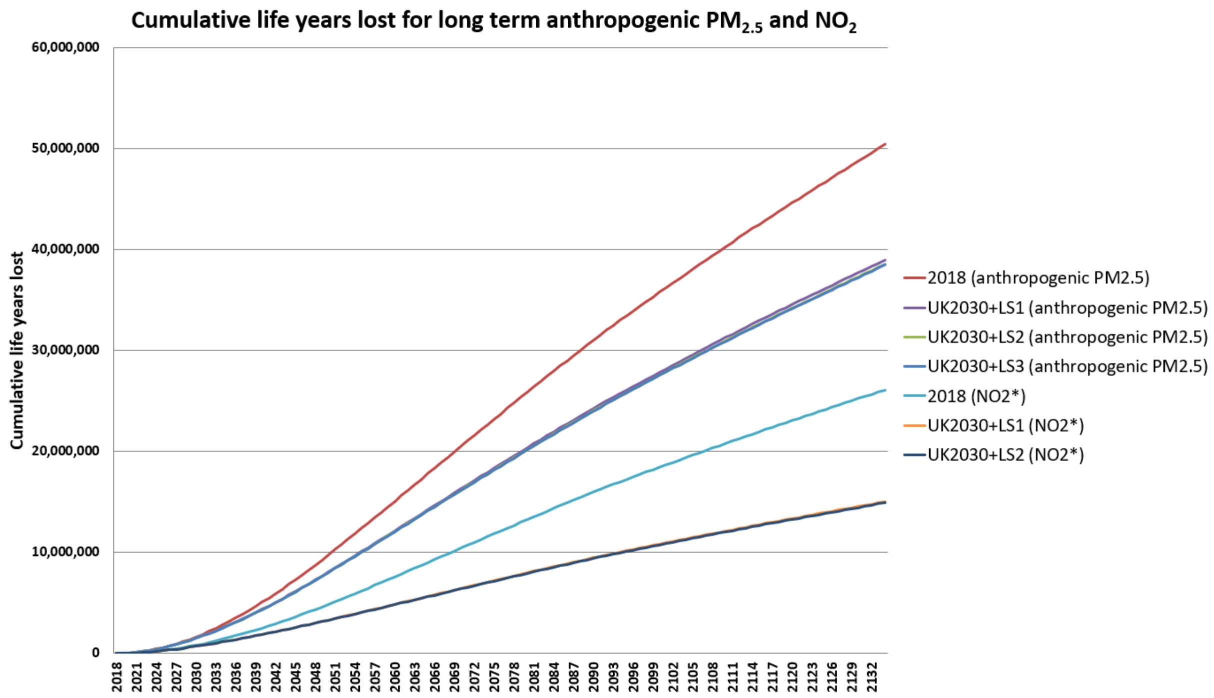
Pollutant	**Scenario**	**Life years lost without cut-off (with cut-off)**		
		Central estimate	**Lower estimate**	**Upper estimate**
Anthropogenic $PM_{2.5}$ (representing the regional air pollution mixture and some of the local mixture)	Baseline: concentration does not reduce from 2018 levels	50,385,911	38,196,091	56,384,559
	Scenario UK2030+LS1: predicted concentration 2018 – 2030 UK2030+LS1	**38,928,722**	29,502,211	43,569,742
	UK2030+LS2: predicted concentration 2018 – 2030 UK2030+LS2	**38,520,008**	29,191,754	43,112,831
	UK2030+LS3: predicted concentration 2018 – 2030 UK2030+LS3	**38,459,099**	29,145,503	43,044,728
NO_2 (representing the local mixture and the rural air pollution hmixture)	Baseline: concentration does not reduce from 2018 levels	26,088,292 (17,858,675)	9,160,576 (6,268,588)	41,600,448 (28,487,556)
	Scenario UK2030+LS1: predicted concentration 2018 - 2030 UK2030+LS1	15,012,116 (7,042,084)	5,266,856 (2,469,940)	23,958,143 (11,241,686)
	Scenario UK2030+LS2: predicted concentration 2018 – 2030 UK2030+LS2	14,907,524 (6,937,240)	-	-
	Scenario UK2030+LS3: predicted concentration 2018 – 2030 UK2030+LS3	14,907,524 (6,937,240)	-	-

For anthropogenic $PM_{2.5}$ assuming no net migration, with projected new births, 2018-2134, compared with life years lived with baseline mortality rates (incorporating mortality improvements over time) with a relative risk (RR) of 1.08 per 10 $\mu g\ m^{-3}$ of anthropogenic $PM_{2.5}$ without cut-off, with lags from the USEPA.

For NO_2 assuming no net migration, with projected new births, 2018-2134, compared with life years lived with baseline mortality rates (incorporating mortality improvements over time) with a relative risk (RR) of 1.023 per 10 $\mu g\ m^{-3}$ of NO_2 without cut-off and with 5 $\mu g\ m^{-3}$ cut-off, with lags from the USEPA.

(Results with cut-offs do not extrapolate beyond the original data, results with no cut-off represent the possibility that there are effects below the cut-off value (it is unknown whether or not this is the case).)

The upper and lower estimates are based on the 95% confidence intervals for the concentration-response functions and not other uncertainties.



**No Cut-off (Cut-off results not shown)*

Figure 20 Cumulative life years lost in the UK for anthropogenic $PM_{2.5}$ and NO_2 if 2018 concentrations remained unchanged and scenario UK2030+LS1, UK2030+LS2 and UK2030+LS3 in 2030 (current and future policies 2018- UK2030+LS1, UK2030+LS2 and UK2030+LS3) across the UK population (no migration) [102]

Figure 20 shows that the cumulative life years lost for the predicted concentrations between 2018 and 2030 accumulates more slowly than the constant 2018 concentration results for both anthropogenic $PM_{2.5}$ and NO_2 as a result of the scenarios reduced concentrations from 2018 to 2030. It is worth remembering that there is a delay before the full benefits of concentration reductions are achieved. This is not just due to a lag between exposure and effect, but also because the greatest gains occur when mortality rates are highest i.e. in the elderly.

Table 29 shows the differences in life years between the predicted concentrations between 2018 and 2030 and both particulate levels and NO_2 concentrations constant at 2018 levels. Using $PM_{2.5}$ as an indicator of the regional pollution and some of the local pollution mixture gives an estimate of 11.5 million life years gained as a result of the predicted concentration changes between 2018 and 2030. Using NO_2 as an indicator of mostly the local pollution mixture and some of the rural pollution gives a slightly lower estimate of 10.8 to 11.1 million life years gained.

[102]With projected new births, compared with life years lived with baseline mortality rates (incorporating mortality improvements over time) 2018-2134. RR 1.08 per 10 μg m^{-3} for anthropogenic $PM_{2.5}$ and RR 1.023 per 10 μg m^{-3} for NO_2, EPA lag. Counterfactual is zero concentrations for NO_2 and non-anthropogenic concentrations for $PM_{2.5}$

The **overall summary** would be that taking into account predicted air pollution concentration changes between 2018 and the 2030 scenario UK2030+LS1, the **UK** population would **gain around 11.5 million life years** over a lifetime.

Table 29 Life years saved 2018-2134 across the UK population of the predicted concentration between 2018 and 2030 (scenario UK2030+LS1, scenario UK2030+LS2 and scenario UK2030+LS3) compared with 2018 anthropogenic $PM_{2.5}$ concentrations and NO_2 remaining unchanged

Pollutant	Scenario	Total life years saved [and percentage] compared with 2018 concentrations maintained without cut-off (with cut-off)		
		Central estimate	Lower estimate	Upper estimate
Anthropogenic $PM_{2.5}$ (representing the regional air pollution mixture and some of the local mixture)	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	**11,457,189 [23%]**	8,693,879	12,814,817
	Scenario UK2030+LS2: Predicted concentration between 2018 and 2030 UK2030+LS2	**11,865,903 [24%]**	9,004,337	13,271,72
	Scenario UK2030+LS3: Predicted concentration between 2018 and 2030 UK2030+LS3	**11,926,812 [24%]**	9,050,588	13,339,830
NO_2 (representing the local mixture and the rural air pollution mixture)	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	11,076,175 (10,816,591)	3,893,720 (3,798,648)	17,642,305 (17,245,870)
	Scenario UK2030+LS2: Predicted concentration between 2018 and 2030 UK2030+LS2	11,180,768 (10,921,436)	-	-
	Scenario UK2030+LS3: Predicted concentration between 2018 and 2030 UK2030+LS3	11,180,768 (10,921,436)	-	-

Note: Figures in bold are the larger of the alternative estimates using $PM_{2.5}$ or NO_2.

Mortality impact (UK countries and Greater Manchester and Glasgow City results)

Table 30 shows the larger results of the two alternatives ($PM_{2.5}$ or NO_2) for the UK by countries and for Greater Manchester and Glasgow City from the life table calculations for anthropogenic $PM_{2.5}$.

Table 30 Total life years lost 2018-2134 across England, Northern Ireland, Scotland and Wales and across Greater Manchester and Glasgow City populations for anthropogenic $PM_{2.5}$ for the 2018 baseline and the 2030 scenario UK2030+LS1

Country / City	**Scenario**	**Life years lost without cut-off**		
		Central estimate	**Lower estimate**	**Upper estimate**
England	Baseline: concentration does not reduce from 2018 levels	44,967,367	34,091,002	50,319,022
	Scenario UK2030+LS1: predicted concentration 2018 – 2030 UK2030+LS1	**34,635,015**	26,249,705	38,763,047
Northern Ireland	Baseline: concentration does not reduce from 2018 levels	1,102,527	835,438	1,234,051
	Scenario UK2030+LS1: predicted concentration 2018 - 2030 UK2030+LS1	**873,646**	661,895	977,949
Scotland	Baseline: concentration does not reduce from 2018 levels	2,409,219	1,824,876	2,697,145
	Scenario UK2030+LS1: predicted concentration 2018 - 2030 UK2030+LS1	**1,962,217**	1,486,186	2,196,802
Wales	Baseline: concentration does not reduce from 2018 levels	1,906,798	1,444,775	2,134,340
	Scenario UK2030+LS1: predicted concentration 2018 - 2030 UK2030+LS1	**1,457,843**	1,104,425	1,631,944
Greater Manchester	Baseline: concentration does not reduce from 2018 levels	2,719,996	2,062,116	3,043,696
	Scenario UK2030+LS1: predicted concentration 2018 - 2030 UK2030+LS1	**2,085,264**	1,580,418	2,333,794

Glasgow City	Baseline: concentration does not reduce from 2018 levels	438,856	332,474	491,259
	Scenario UK2030+LS1: predicted concentration 2018 - 2030 UK2030+LS1	**350,857**	265,777	392,774

For anthropogenic $PM_{2.5}$ assuming no net migration, with projected new births, 2018-2134, compared with life years lived with baseline mortality rates (incorporating mortality improvements over time) with a relative risk (RR) of 1.08 per 10 µg m^{-3} of anthropogenic $PM_{2.5}$ without cut-off, with lags from the USEPA.

The upper and lower estimates are based on the 95% confidence intervals for the concentration-response functions and not other uncertainties.

Table 31 shows the differences in life years between the predicted concentrations between 2018 and 2030 scenario UK2030+LS1 and particulate levels constant at 2018 levels. A summary would be that taking into account predicted air pollution concentration changes between 2018 and 2030 UK2030+LS1, the population in England, Northern Ireland and Scotland/Wales would gain around 10.3 million, 230,000 and 450,000 life years over a lifetime, respectively. For **Greater Manchester**, the population would gain around **630,000 life years** over a lifetime compared with a gain around **90,000 life years** for **Glasgow City**.

Table 31 Life years saved across England, Northern Ireland, Scotland and Wales population of the predicted concentration between 2018 and 2030 scenario UK2030+LS1 compared with 2018 anthropogenic $PM_{2.5}$ concentrations remaining unchanged

Country / City	**Scenario**	**Total life years saved [and percentage] compared with 2018 concentrations maintained without cut-off**		
		Central estimate	**Lower estimate**	**Upper estimate**
England	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	**10,332,352 [23%]**	7,841,297	11,555,975
Northern Ireland	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	**228,880 [21%]**	173,543	256,103
Scotland	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	**447,001 [19%]**	338,690	500,343
Wales	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	**448,955 [24%]**	340,350	502,396
Greater Manchester	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	**634,732 [23%]**	481,698	709,902
Glasgow City	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	**87,999 [20%]**	66,696	98,485

Note: Greater Manchester formed of ten local authorities (Bolton, Bury, Manchester, Oldham, Rochdale, Salford, Stockport, Tameside, Trafford and Wigan)

10.2. London mortality impacts

Table 32 shows London results from the life table calculations for anthropogenic $PM_{2.5}$ and NO_2 assuming (i) that the concentration does not reduce from 2018 levels or (ii) that the predicted concentrations changed between 2018 and 2030 scenario LS1, LS2 and LS3.

In summary, if 2018 concentrations of $PM_{2.5}$ remained unchanged for 117 years, around 8.44 million life years would be lost across London's population over that period. This improves to around 6.44, 6.03 and 5.97 million life years lost for London with the predicted concentration changes between 2018 and 2030 scenario LS1, LS2 and LS3, respectively.

Table 32 Total life years lost across London population for anthropogenic $PM_{2.5}$ and NO_2 for the 2018 baseline and the 2030 scenario LS1, LS2 and LS3

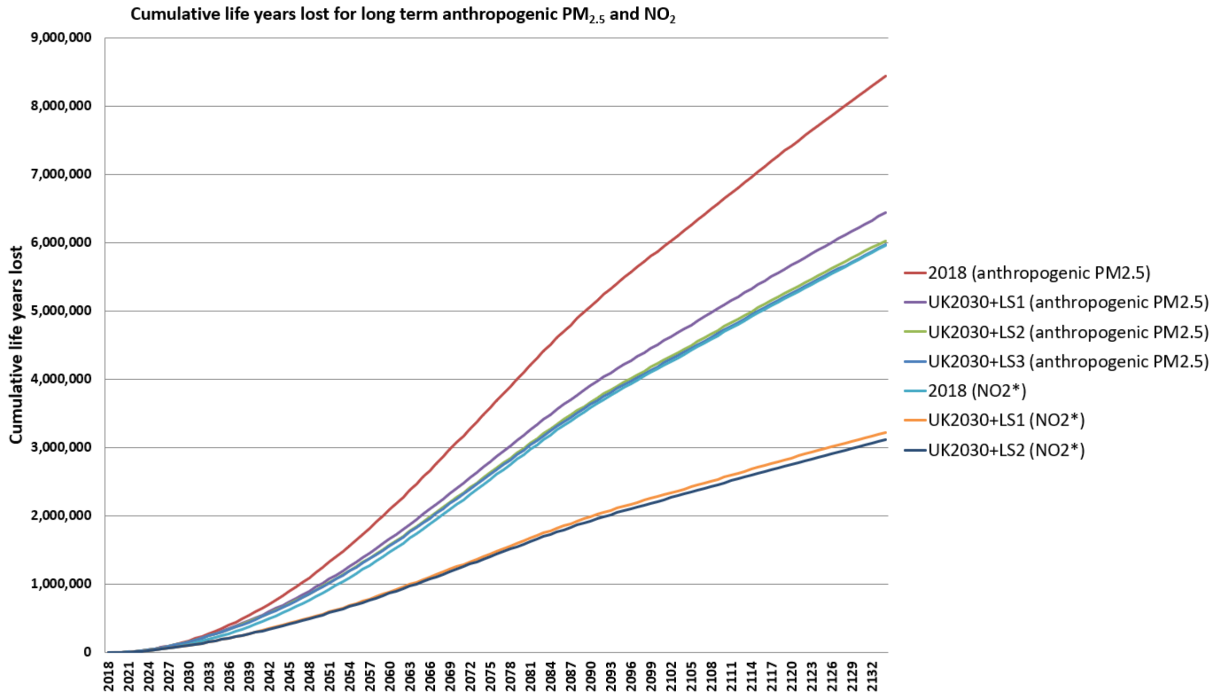
Pollutant	Scenario	Life years lost without cut-off (with cut-off)		
		Central estimate	**Lower estimate**	**Upper estimate**
Anthropogenic $PM_{2.5}$ (representing the regional air pollution mixture and some of the local mixture)	Baseline: concentration does not reduce from 2018 levels	8,440,302	6,404,043	9,440,953
	Scenario LS1: predicted concentration 2018 – 2030 LS1	**6,439,358**	4,883,275	7,204,698
	Scenario LS2: predicted concentration 2018 – 2030 LS2	**6,030,644**	4,572,818	6,747,786
	Scenario LS3: predicted concentration 2018 – 2030 LS3	**5,969,736**	4,526,567	6,679,684
NO_2 (representing the local mixture and the rural air pollution mixture)	Baseline: concentration does not reduce from 2018 levels	5,957,354 (4,785,889)	2,096,058 (1,682,606)	9,481,293 (7,622,575)
	Scenario LS1: predicted concentration 2018 – 2030 LS1	3,224,612 (2,046,515)	1,132,480 (718,243)	5,141,292 (3,265,067)
	Scenario LS2: predicted concentration 2018 – 2030 LS2	3,120,019 (1,941,670)	-	-
	Scenario LS3: predicted concentration 2018 – 2030 LS3	3,120,019 (1,941,670)	-	-

For anthropogenic $PM_{2.5}$ assuming no net migration, with projected new births, 2018-2134, compared with life years lived with baseline mortality rates (incorporating mortality improvements over time) with a relative risk (RR) of 1.08 per 10 µg m^{-3} of anthropogenic $PM_{2.5}$ without cut-off, with lags from the USEPA.

For NO_2 assuming no net migration, with projected new births, 2018-2134, compared with life years lived with baseline mortality rates (incorporating mortality improvements over time) with a relative risk (RR) of 1.023 per 10 µg m^{-3} of NO_2 without cut-off and with 5 µg m^{-3} cut-off, with lags from the USEPA.

(Results with cut-offs do not extrapolate beyond the original data, results with no cut-off represent the possibility that there are effects below the cut-off value (it is unknown whether or not this is the case).)

The upper and lower estimates are based on the 95% confidence intervals for the concentration-response functions and not other uncertainties.



*No Cut-off (Cut-off results not shown)

Figure 21 Cumulative life years lost in London for anthropogenic $PM_{2.5}$ and NO_2 if 2018 concentrations remained unchanged and scenario LS1, LS2 and LS3 in 2030 (current and future policies 2018-2030 UK2030+LS1, UK2030+LS2 and UK2030+LS3) across London population (no migration) [103]

Figure 21 shows that the cumulative life years lost for the predicted concentration between 2018 and 2030 accumulates more slowly than the constant 2018 concentration results for both anthropogenic $PM_{2.5}$ and NO_2 as a result of the scenarios reduced concentrations from 2018 to 2030. It is worth remembering that there is a delay before the full benefits of concentration reductions are achieved. This is not just due to a lag between exposure and effect, but also because the greatest gains occur when mortality rates are highest i.e. in the elderly.

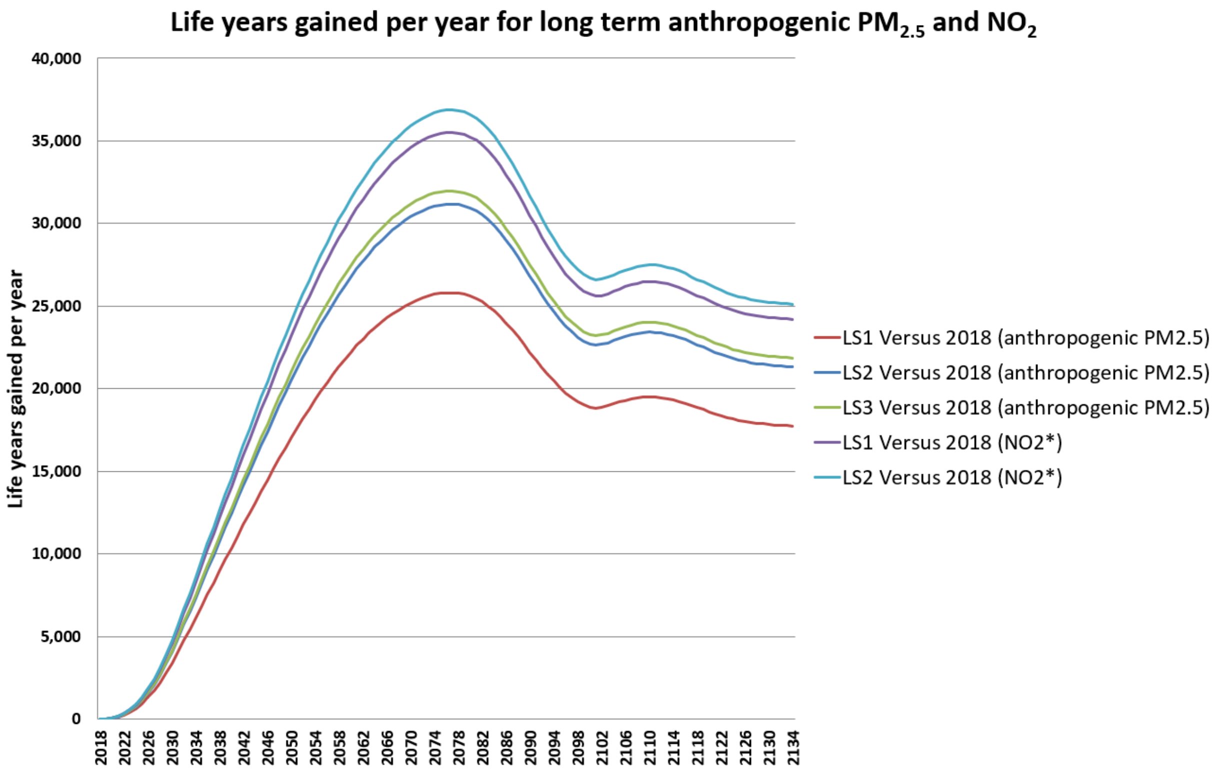
Table 33 shows the differences in London between the predicted concentrations between 2018 and 2030 (LS1, LS2 and LS3) levels and particulate matter concentrations constant at 2018. With **scenario LS1**, the population in **London** would **gain** around **2 million life years** over a lifetime compared with a gain around 2.4 million life years for LS2 and a gain around 2.5 million life years for LS3.

[103]*With projected new births, compared with life years lived with baseline mortality rates (incorporating mortality improvements over time) 2018-2134. RR 1.08 per 10 µg m^{-3} for anthropogenic $PM_{2.5}$ and RR 1.023 per 10 µg m^{-3} for NO_2, EPA lag. Counterfactual is zero concentrations for NO_2 and non-anthropogenic concentrations for $PM_{2.5}$*

Table 33 Life years saved from 2018-2134 across the London population from the predicted concentration between 2018 and 2030 scenario LS1, LS2 and LS3 compared with 2018 anthropogenic $PM_{2.5}$ concentrations and NO_2 remaining unchanged

Pollutant	**Scenario**	**Total life years saved [and percentage] compared with 2018 concentrations maintained without cut-off (with cut-off)**		
		Central estimate	**Lower estimate**	**Upper estimate**
Anthropogenic $PM_{2.5}$ (representing the regional air pollution mixture and some of the local mixture)	Scenario LS1: Predicted concentration between 2018 and 2030 LS1	2,000,944 [24%]	1,520,767	2,236,255
	Scenario LS2: Predicted concentration between 2018 and 2030 LS2	2,409,658 [28.5%]	1,831,225	2,693,166
	Scenario LS3: Predicted concentration between 2018 and 2030 LS3	2,470,567 [29.3%]	1,877,476	2,761,269
NO_2 (representing the local mixture and the rural air pollution mixture)	Scenario LS1: Predicted concentration between 2018 and 2030 LS1	2,732,742 (**2,739,374**)	963,578 (964,363)	4,340,001 (4,357,508)
	Scenario LS2: Predicted concentration between 2018 and 2030 LS2	2,837,334 (**2,844,219**)	-	-
	Scenario LS3: Predicted concentration between 2018 and 2030 LS3	2,837,334 (**2,844,219**)	-	-

Note: Figures in bold are the larger of the alternative estimates using $PM_{2.5}$ or NO_2



*No Cut-off (Cut-off results not shown)

Figure 22 Life years gained per year from long-term exposure to the improvements in pollution from 2018 to 2030 (scenario LS1, LS2 and LS3) of anthropogenic $PM_{2.5}$ and NO_2 relative to 2018 concentrations remaining unchanged

Figure 22 shows the effect in London of the decrease in anthropogenic $PM_{2.5}$ and NO_2 concentration from 2018 to 2030 for scenario LS1, LS2 and LS3. The gains are greater for NO_2, despite the smaller concentration-response function for NO_2 (NO_2 crf 1.023 and $PM_{2.5}$ 1.08) and mortality, due to the larger concentration reductions of NO_2 compared with $PM_{2.5}$ in London between 2018 and 2030.

10.3. UK Life-expectancy from birth in 2018

Total life years across the population is the most appropriate metric for cost-benefit analysis of policies as it captures effects in the entire population. However, it is a difficult type of metric to communicate as it is difficult to judge what is a ‘small’ answer or a ‘large’ answer. Life-expectancy from birth is a more familiar concept for the general public, although it only captures effects on those born on a particular date. Results for life expectancy from birth are shown for the UK in Table 34, by countries including Greater Manchester and Glasgow City in Table 35 and for London in Table 36.

The average loss of life expectancy from birth in the UK would be about 35 weeks for males and 31 weeks for females if 2018 $PM_{2.5}$ concentrations were unchanged but improves to 27 weeks for males and 24 weeks for females for the predicted concentration changes between 2018 and 2030 scenario UK2030+LS1 (an improvement by about 8-9 weeks).

Using NO_2, the average loss of life expectancy from birth in London would be about 13 – 19 weeks for males and 11 – 16 weeks for females if NO_2 concentrations were unchanged from 2018 but improves by about 7 – 8 weeks to 5 – 10 weeks for males and 4 – 9 weeks for females with projected future changes between 2018 and 2030 included scenario UK2030+LS1.

A summary would be that the UK projected future changes in air pollution concentrations up to 2030 scenario UK2030+LS1 (projected from 2018) provide an improvement in average life expectancy (from birth in 2018) of around 2 months but an average loss of life expectancy (from birth in 2018) of around 5.5–6 months remains even with the reduced concentrations. Males are more affected than females – this is mainly due to the higher mortality rates in men compared with women rather than differences in air pollution exposure. The concentration-response function is implemented as a percentage change in baseline mortality rates. If the baseline mortality rates are higher, then the absolute impact is higher even though the percentage change is the same.

Table 34 Loss of life expectancy by gender across UK from birth in 2018 (followed for 105 years) for anthropogenic $PM_{2.5}$ and NO_2

Pollutant	**Scenario**	**Loss of life expectancy from birthcompared with baseline mortality rates, 2018 birth cohort (in weeks)** **without cut-off** **(with cut-off)**		**Gain [and percentage] of life expectancy from birth compared with baseline mortality rates, 2018 birth cohort (in weeks)** **without cut-off** **(with cut-off)**	
		Male	**Female**	**Male**	**Female**
Anthropogenic $PM_{2.5}$	Concentration does not reduce from 2018 levels	35.19	31.03	-	-
	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	**26.66**	**23.52**	8.53 [24%]	7.52 [24%]
	Scenario UK2030+LS2: Predicted concentration between 2018 and 2030 UK2030+LS2	**26.33**	**23.24**	8.86 [25%]	7.80 [25%]
	Scenario UK2030+LS3: Predicted concentration between 2018 and 2030 UK2030+LS3	**26.28**	**23.19**	8.91 [25%]	7.84 [25%]
NO_2	Concentration does not reduce from 2018 levels	18.57 (12.89)	16.29 (11.26)	-	-
	Scenario UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	10.17 (4.66)	8.93 (4.05)	8.40 [45%] (8.23 [64%])	7.36 [45%] (7.21 [64%])
	Scenario UK2030+LS2: Predicted concentration between 2018 and 2030 UK2030+LS2	10.09 (4.57)	8.85 (3.97)	8.48 [46%] (8.32 [65%])	7.44 [46%] (7.29 [65%])
	Scenario UK2030+LS3: Predicted concentration between 2018 and 2030 UK2030+LS3	10.09 (4.57)	8.85 (3.97)	8.48 [46%] (8.32 [65%])	7.44 [46%] (7.29 [65%])

Note Figures in bold are the larger of the alternative estimates using $PM_{2.5}$ or NO_2

Life-expectancy from birth in 2018 (UK countries including Greater Manchester and Glasgow City results)

A summary would be that England, Northern Ireland, Scotland and Wales and, Greater Manchester and Glasgow City projected future changes in air pollution concentrations up to 2030 scenario UK2030+LS1 (projected from 2018) provide an improvement in average life expectancy (from birth in 2018) of around 1 to 2.5 months, but an average loss of life expectancy (from birth in 2018) of around 3.5 to 7.5 months remains even with the reduced concentrations.

Table 35 Loss of life expectancy by gender across England, Northern Ireland, Scotland and Wales and, Greater Manchester and Glasgow City results from birth in 2018 (followed for 105 years) for anthropogenic $PM_{2.5}$

Pollutant	**Scenario**	**Loss of life expectancy from birth compared with baseline mortality rates, 2018 birth cohort (in weeks) without cut-off**		**Gain [and percentage] of life expectancy from birth compared with baseline mortality rates, 2018 birth cohort (in weeks) without cut-off**	
		Male	**Female**	**Male**	**Female**
England	Concentration does not reduce from 2018 levels	36.9	32.6	-	-
	UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	27.9	24.6	9.0 [24%]	8.0 [24%]
Northern Ireland	Concentration does not reduce from 2018 levels	26.3	23.6	-	-
	UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	20.5	18.4	5.8 [22%]	5.2 [22%]
Scotland	Concentration does not reduce from 2018 levels	21.9	19.1	-	-
	UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	17.5	15.3	4.4 [20%]	3.8 [20%]
Wales	Concentration does not reduce from 2018 levels	29.2	25.6	-	-

	UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	21.8	19.1	7.4 [25%]	6.5 [25%]
Greater Manchester	Concentration does not reduce from 2018 levels	42.5	37.2	-	-
	UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	32	28	10.5 [25%]	9.2 [25%]
Glasgow City	Concentration does not reduce from 2018 levels	33	28	-	-
	UK2030+LS1: Predicted concentration between 2018 and 2030 UK2030+LS1	25.9	22	7.1 [21%]	6 [21%]

10.4. London Life-expectancy from birth in 2018

The **overall summary** would be that **London** projected future changes in air pollution concentrations up to 2030 scenario (projected from 2018) provide an **improvement** in average **life expectancy** (from birth in 2018) **of around 2–2.5 and 2.5–3 months** for 2030 scenario LS1 and LS2/ LS3, respectively, but an average loss of life expectancy (from birth in 2018) of around 6-7 and 5.5-6.5 months for 2030 scenario LS1 and LS2/ LS3, respectively, still remains even with the reduced concentrations.

Table 36 Loss of life expectancy by gender across London from birth in 2018 (followed for 105 years) for anthropogenic $PM_{2.5}$ and NO_2

Pollutant	**Scenario**	**Loss of life expectancy from birth compared with baseline mortality rates, 2018 birth cohort (in weeks) without cut-off (with cut-off)**		**Gain [and percentage] of life expectancy from birth compared with baseline mortality rates, 2018 birth cohort (in weeks) without cut-off (with cut-off)**	
		Male	**Female**	**Male**	**Female**
	Concentration does not reduce from 2018 levels	40.1	34.7	-	-

Anthropogenic $PM_{2.5}$	Scenario LS1: Predicted concentration between 2018 and 2030 LS1	**30.1**	**26.1**	9.9 [25%]	8.6 [25%]
	Scenario LS2: Predicted concentration between 2018 and 2030 LS2	**28.1**	**24.4**	11.9 [30%]	10.3 [30%]
	Scenario LS3: Predicted concentration between 2018 and 2030 LS3	**27.8**	**24.1**	12.2 [31%]	10.6 [30%]
h NO_2	Concentration does not reduce from 2018 levels	28.2 (22.6)	24.4 (19.5)	-	-
	Scenario LS1: Predicted concentration between 2018 and 2030 LS1	14.7 (9.1)	12.7 (7.8)	13.5 [48%] (13.6 [60%])	11.7 [48%] (11.7 [60%])
	Scenario LS2: Predicted concentration between 2018 and 2030 LS2	14.2 (8.5)	12.3 (7.4)	14.0 [50%] (14.1 [62%])	12.1 [50%] (12.1 [62%])
	Scenario LS3: Predicted concentration between 2018 and 2030 LS3	14.2 (8.5)	12.3 (7.4)	14.0 [50%] (14.1 [62%])	12.1 [50%] (12.1 [62%])

Note Figures in bold are the larger of the alternative estimates using $PM_{2.5}$ or NO_2

10.5. Other health outcomes - UK

In addition to the gains in life expectancy, there are benefits from reductions in a range of other health effects. These include improvements in both lung and heart disease effects and improvements in both symptoms and incidence of new disease. .

The average concentrations used as an approximate input for the health outcomes other than gains in life years are given in *Table 37*. For $PM_{2.5}$ the results are very similar to the population-weighted average concentrations in *Table 25*. For NO_2, the average concentrations are a small underestimate of the population-weighted average concentrations given in *Table 27*.

Table 37 UK average concentrations (µg m^{-3}) in 2018 and 2030 and the difference between them (average from 20x20m to ward and average from ward to UK)

	2018	UK2030+LS1	UK2030+LS2	UK2030+LS3
UK				
NO_2	12.959	7.176	7.140	
Total PM_{10}	15.197	13.139	13.097	13.091
Total $PM_{2.5}$	9.027	6.967	6.924	6.919
Anthropogenic PM_{10}	11.162	9.039	8.997	8.991
Anthropogenic $PM_{2.5}$	8.457	6.406	6.364	6.358
NO_2 change VS 2018		-5.784	-5.819	
PM_{10} change VS 2018		-2.058	-2.100	-2.106
$PM_{2.5}$ change VS 2018		-2.060	-2.102	-2.108
London				
NO_2	25.751	13.251	12.768	
Total PM_{10}	17.311	14.832	14.257	14.177
Total $PM_{2.5}$	11.262	8.551	7.977	7.898
Anthropogenic PM_{10}	13.536	10.977	10.402	10.322
Anthropogenic $PM_{2.5}$	10.711	8.020	7.446	7.367
NO_2 change		-12.500	-12.982	
PM_{10} change		-2.480	-3.054	-3.134
$PM_{2.5}$ change		-2.711	-3.284	-3.364

Scenario UK2030+LS1

Across the UK for scenario UK2030+LS1 compared with 2018 levels remaining unchanged, the number of symptom days in asthmatic children is projected to reduce by 388,000 on average each year and the number of new cases of coronary heart disease is projected to reduce by 3100 each year(Table 38, Figure 23, Figure 24). There are also other health outcomes affected for both respiratory and cardiovascular endpoints. The numbers for chronic phlegm are quite large at 149,000 cases of chronic phlegm symptoms on average per year but also uncertain (see section on chronic phlegm in Chapter 7 on health impact assessment methods). Cardiovascular admissions are one of the smaller outcomes in terms of case numbers (2,700) but they are more serious than symptom days. This is reflected in the different ranking of health outcomes after monetary valuation has been applied, since this provides some indication of severity (cross-reference monetary valuation section). This is even more the case for infant deaths which are fortunately rare and understandably have a high valuation. They are also more uncertain given the evidence on which the calculations are based (Table 11). However, even given the caveats, there are clearly a wide range of health benefits occurring as a result of the air pollution reductions.

Table 38 Reductions in health effects (average cases per year UK) from air pollution reductions for Scenario UK2030+LS1 compared with 2018 concentrations remaining unchanged

Health effect	**Reduction in average cases per year**		
	Central estimate	**Lower estimate**	**Upper estimate**
Asthmatic symptom days in asthmatic children (PM_{10})	388,018	84,176	697,867
Chronic phlegm in adults (PM_{10})	148,757	10,854	280, 697
Chronic bronchitic symptoms in asthmatic children (NO_2)	24,916	12,305	67,674
Acute bronchitis infections in children (PM_{10})	12,937	3,416	29,057
Respiratory hospital admissions (NO_2)*	3,655	2,120	5,248
New cases coronary heart disease ($PM_{2.5}$)	3,077	477	6,675
Cardiovascular hospital admissions (NO_2)*	2,689	1,307	4,103
Infant deaths (PM_{10})	23	12	40

*Decreases in hospital admissions were calculated for both PM2.5 and NO2 and the largest result between the two pollutants taken (summing both would lead to double counting due to overlap between the pollutants).

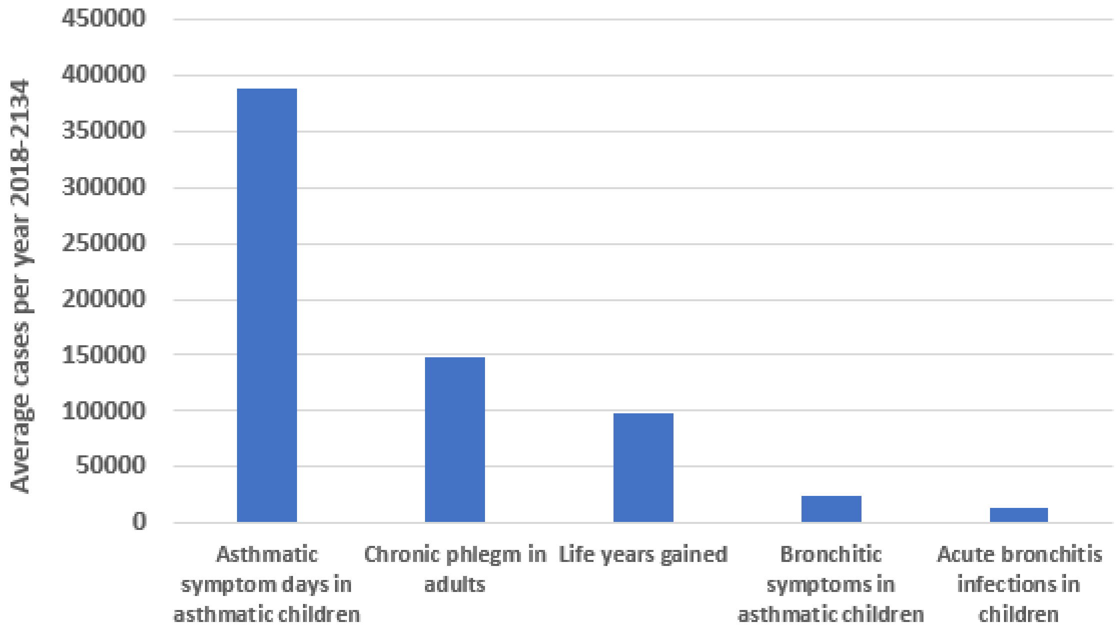


Figure 23 Reductions in respiratory symptoms and infections (average per year UK) from air pollution reductions Scenario UK2030+LS1 compared with 2018 concentrations remaining unchanged

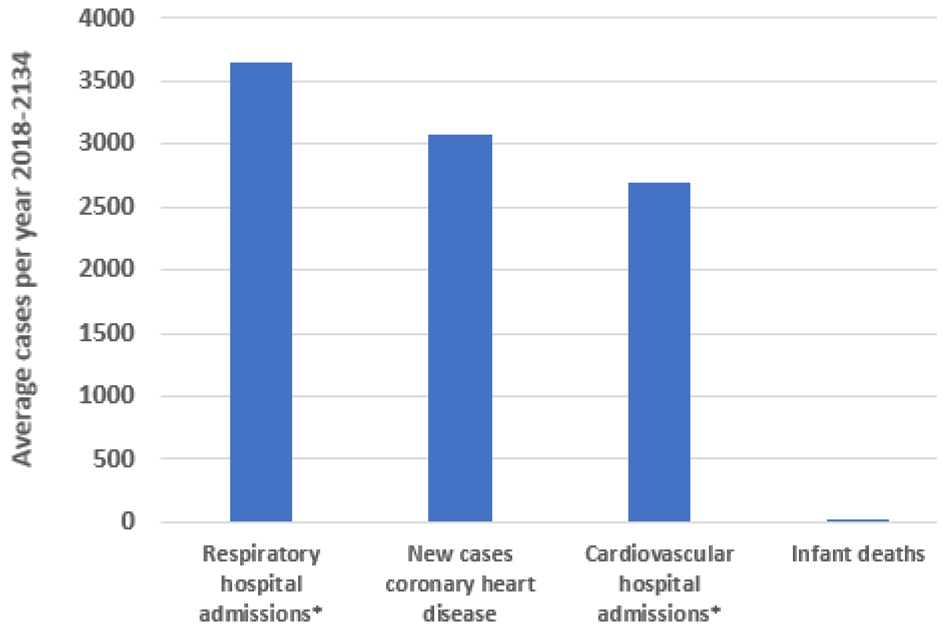


Figure 24 Reductions in hospital admissions, new cases of coronary heart disease and infant deaths (average cases per year UK) Scenario UK2030+LS1 compared with 2018 concentrations remaining unchanged. (Note difference in scale compared with Figure 23).

10.6. Other health outcomes - London

Scenario UK2030+LS2

The total health benefits from Scenario UK2030+LS2 compared with 2018 levels remaining unchanged (Table 39, Figure 25, Figure 26) are increased compared with Scenario UK2030+LS1. This is not that clear in the figures as Scenario UK2030+LS1 provides a large proportion of the benefits within Scenario UK2030+LS2 as well (see next paragraph).

Table 39 Reductions in health effects (average cases per year UK) from air pollution reductions for Scenario UK2030+LS2 compared with 2018 concentrations remaining unchanged

Health effect	Reduction in average cases per year		
	Central estimate	**Lower estimate**	**Upper estimate**
Asthmatic symptom days in asthmatic children (PM_{10})	396,013	85,912	712,228
Chronic phlegm in adults (PM_{10})	151,737	11,079	286,209
Chronic bronchitic symptoms in asthmatic children (NO_2)	25,067	12,381	68,069
Acute bronchitis infections in children (PM_{10})	13,203	3,486	29,648
Respiratory hospital admissions (NO_2)*	3,677	2,133	5,280
New cases coronary heart disease ($PM_{2.5}$)	3,121	477	6,805
Cardiovascular hospital admissions (NO_2)*	2,705	1,315	4,128
Infant deaths (PM_{10})	23	12	40

*Decreases in hospital admissions were calculated for both PM2.5 and NO2 and the largest result between the two pollutants taken (summing both would lead to double counting due to overlap between the pollutants).

The additional benefits comparing UK 2030 +LS2 with scenario UK2030+LS1 occur only in London.

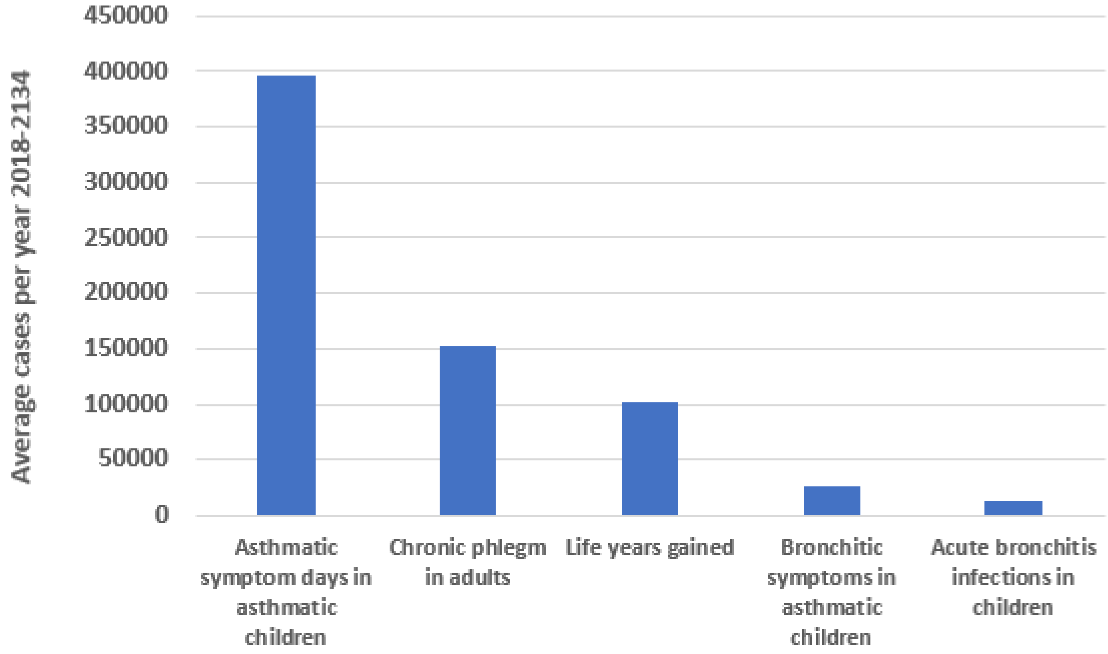


Figure 25 Reductions in respiratory symptoms and infections (average cases per year UK) from air pollution reductions for UK 2030 + LS2 compared with 2018 concentrations remaining unchanged

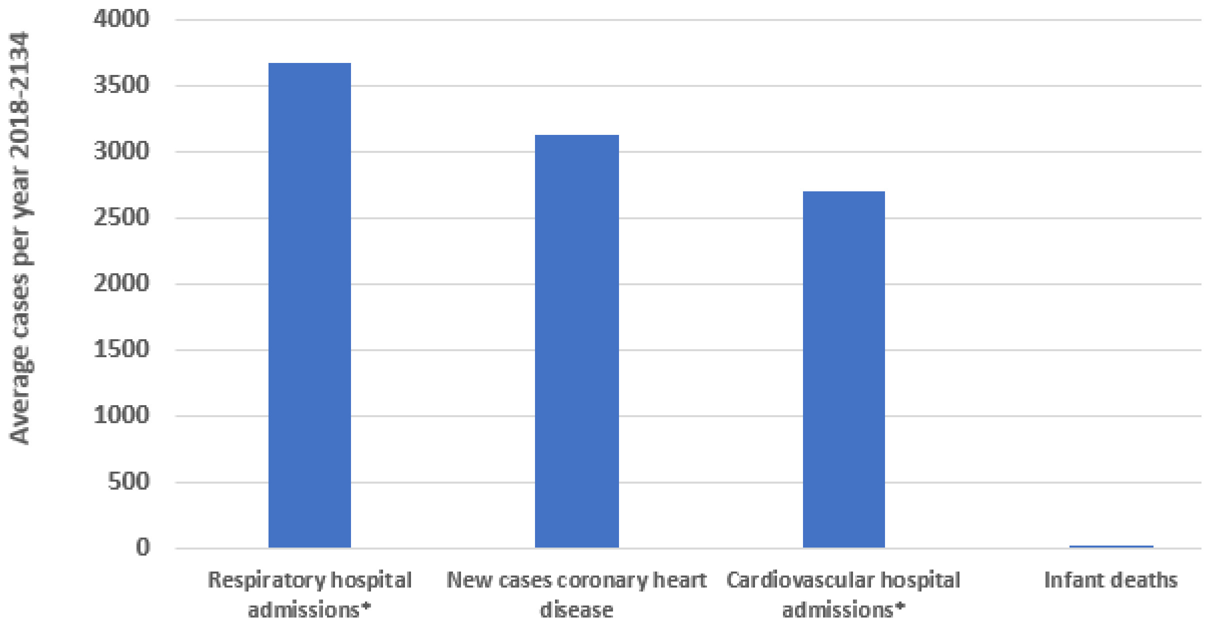


Figure 26 Reductions in hospital admissions, new cases of coronary heart disease and infant deaths (average cases per year UK) Scenario UK2030+LS2 compared with 2018 concentrations remaining unchanged. (Note difference in scale compared with Figure 25).

The additional benefits from Scenario UK2030+LS2 compared with Scenario UK2030+LS1 (as opposed to the overall benefits of Scenario UK2030+LS2 which includes Scenario UK2030+LS1) are given in Table 40.The additional benefits of Scenario UK2030+LS2 are smaller but they are concentrated in London. The proportional increase is slightly greater for $PM_{2.5}$ and PM_{10} related outcomes compared with those for NO_2. This is not so much to do with the additional policies in Scenario UK2030+LS2 as it is to do with the large reductions in NO_2 from electrification of the fleet in Scenario UK2030+LS1. No change is shown for infant deaths, not because PM_{10} does not change but because infant deaths are fortunately rare.

Table 40 Reductions in health effects (average cases per year UK) from air pollution reductions for Scenario UK2030+LS2 compared with Scenario UK2030+LS1

	Reduction in average cases per year
Health effect	**Central estimate**
Asthmatic symptom days in asthmatic children (PM_{10})	7995
Chronic phlegm in adults (PM_{10})	2981
Chronic bronchitic symptoms in asthmatic children (NO_2)	151
Acute bronchitis infections in children (PM_{10})	266
Respiratory hospital admissions (NO_2)*	22
New cases coronary heart disease ($PM_{2.5}$)	43
Cardiovascular hospital admissions (NO_2)*	17
Infant deaths (PM_{10})	0

*Decreases in hospital admissions were calculated for both PM2.5 and NO2 and the largest result between the two pollutants taken (summing both would lead to double counting due to overlap between the pollutants)

Scenario UK2030+LS3

The total health benefits from Scenario UK2030+LS3 compared with 2018 levels remaining unchanged are slightly further increased compared with Scenarios UK2030+LS1 and (Table 41, Figure 27, Figure 28) for at least some outcomes. This is not that clear in the figures as Scenario UK2030+LS1 provides a large proportion of the benefits within Scenario UK2030 +LS3 as well.

Table 41 Reductions in health effects (average cases per year UK) from air pollution reductions for Scenario UK2030+LS3 compared with 2018 concentrations remaining unchanged

Health effect	Reduction in average cases per year		
	Central estimate	**Lower estimate**	**Upper estimate**
Asthmatic symptom days in asthmatic children (PM_{10})	397,123	86,154	714,221
Chronic phlegm in adults (PM_{10})	152,187	11,136	287,150
Chronic bronchitic symptoms in asthmatic children (NO_2)	25,067	12,381	68,069
Acute bronchitis infections in children (PM_{10})	13,241	3,497	29,730
Respiratory hospital admissions (NO_2)*	3,677	2,133	5,280
New cases coronary heart disease ($PM_{2.5}$)	3,121	477	6,805
Cardiovascular hospital admissions (NO_2)_	2,705	1,315	4,128
Infant deaths (PM_{10})	24	12	41

*Decreases in hospital admissions were calculated for both $PM_{2.5}$ and NO_2 and the largest result between the two pollutants taken (summing both would lead to double counting due to overlap between the pollutants)

The additional benefits comparing UK 2030 +LS2 with scenario UK2030+LS1 occur only in London.

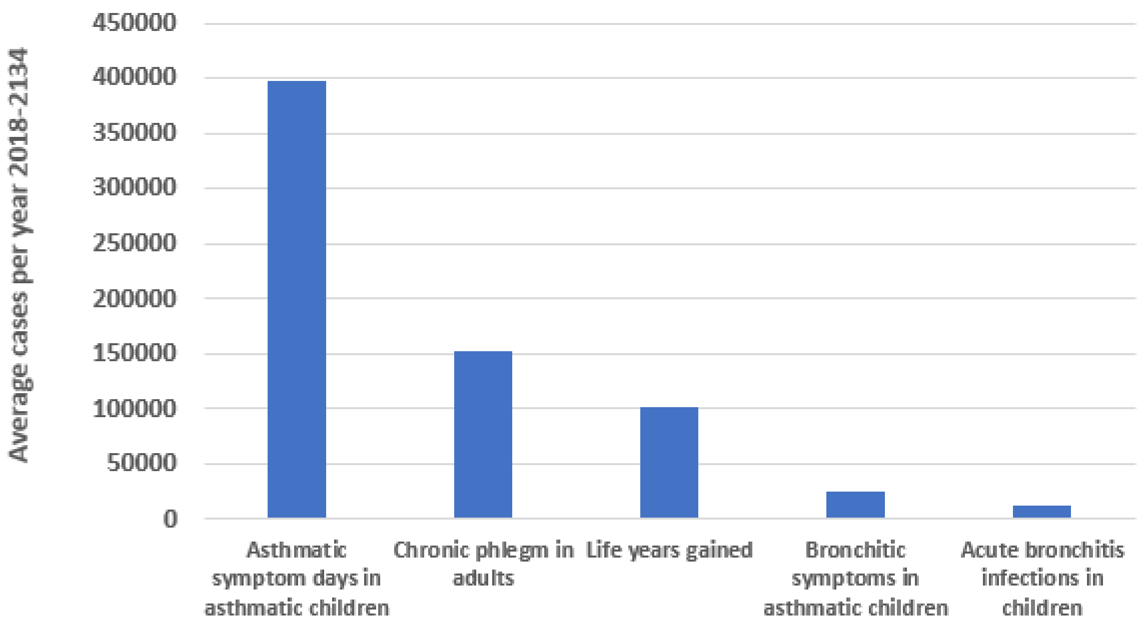


Figure 27 Reductions in respiratory symptoms and infections (average cases per year UK) from air pollution reductions for Scenario UK2030+LS3 compared with 2018 concentrations remaining unchanged

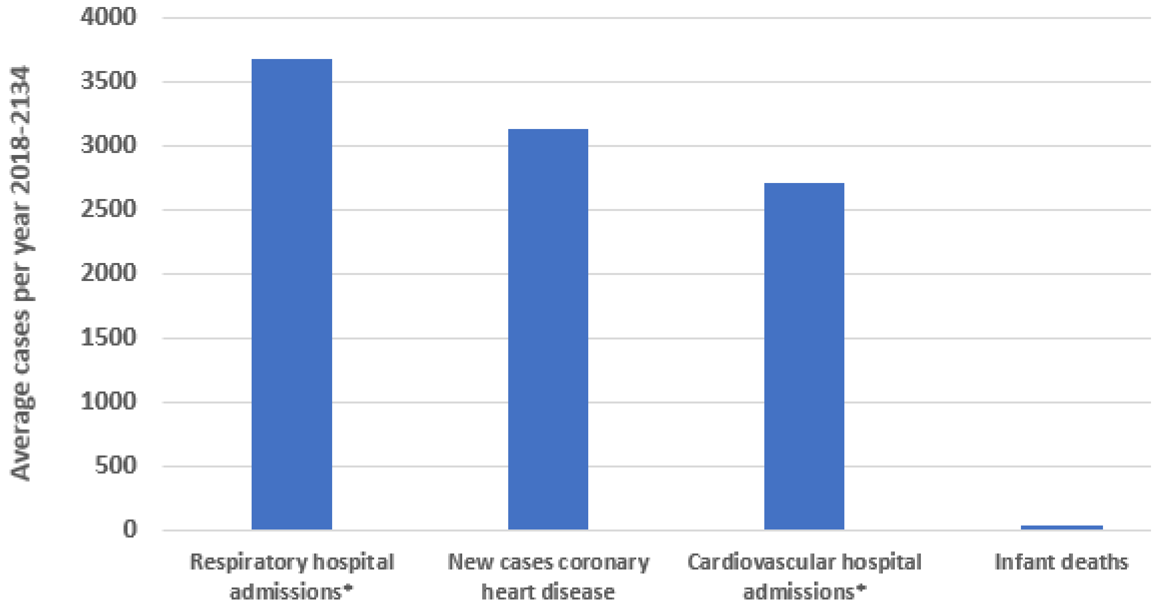


Figure 28 Reductions in hospital admissions, new cases of coronary heart disease and infant deaths (average cases per year UK) Scenario UK2030+LS3 compared with 2018 concentrations remaining unchanged. (Note difference in scale compared with Figure 27)

The additional benefits from Scenario UK2030+LS3 compared with Scenario UK2030+LS2 (as opposed to the overall benefits of Scenario UK2030+LS3 which includes Scenario UK2030+LS1 and UK2030+LS2) are given in Table 42. The additional benefits of Scenario UK2030+LS3 are small because not only do they only occur in London but they only address one sector (domestic wood burning). (Of course, the costs are likely to be smaller too, although they are not derived here because Scenario UK2030+LS3 is a general ambition rather than a precisely defined policy). Some further health outcomes related to particular matter show no change because the outcomes are too rare to be affected above the level of rounding (e.g. new cases of coronary heart disease).

There are no additional benefits for NO_2 because NO_2 concentrations are not decreased for Scenario UK2030+LS3 compared with Scenario UK2030+LS2. Due to overlap in the effects of NO_2 and $PM_{2.5}$, numbers for respiratory and cardiovascular hospital admissions were only given for the pollutant with the largest result, For the overall benefits, this is always NO_2 which, as mentioned previously, gives large changes for Scenario UK2030+LS1. For the difference between Scenario UK2030+LS3 and UK2030+LS2, this is no longer the case. So, the results for $PM_{2.5}$ and hospital admissions are given in Table 43 (with results for the other scenarios as well for completeness).

Table 42 Reductions in health effects (average cases per year UK) from air pollution reductions for Scenario UK2030+LS3 compared with Scenario UK2030+LS2

	Reduction in average cases per year
Health effect	**Central estimate**
Asthmatic symptom days in asthmatic children (PM_{10})	1110
Chronic phlegm in adults (PM_{10})	450
Chronic bronchitic symptoms in asthmatic children (NO_2)	0
Acute bronchitis infections in children (PM_{10})	38
Respiratory hospital admissions (NO_2)*	0
New cases coronary heart disease ($PM_{2.5}$)	0
Cardiovascular hospital admissions (NO_2)*	0
Infant deaths (PM_{10})	0

*Decreases in hospital admissions were calculated for both PM2.5 and NO2 and the largest result between the two pollutants taken (summing both would lead to double counting due to overlap between the pollutants). In this case, the result for PM2.5 related hospital admissions should be substituted here see Table 43 below.

Table 43 Reductions in $PM_{2.5}$ related hospital admissions (average cases per year UK) from air pollution reductions for Scenario UK2030+LS3 compared with Scenario UK2030+LS2

	Reduction in $PM_{2.5}$ related hospital admissions[a] (average cases per year)				
Health effect	**Scenario UK2030+LS1 vs 2018**	**Scenario UK2030+LS2 vs 2018**	**Scenario UK2030+LS3 vs 2018**	**Scenario UK2030+LS2 vs UK2030+LS1**	**Scenario UK2030+LS3 vs UK2030+LS2**
Respiratory hospital admissions ($PM_{2.5}$)	2189	2235	2241	46	6
Cardiovascular hospital admissions ($PM_{2.5}$)	1305	1332	1336	27	4

[a] Alternative figures for the hospital admission numbers in the preceding tables. (As the results for $PM_{2.5}$ and NO_2 overlap to some extent we have used only one of the alternative results – choosing the highest result. This is usually NO_2 for all scenarios except for the difference between the results for Scenario UK2030+LS3 and Scenario UK2030+LS2.)

10.7. Discussion

The results in this chapter show substantial health benefits, particularly for the UK2030+LS1 scenario (11.5 million life years from 2018 – 2134). A further 0.4 and 0.5 million life years are gained for the UK2030+LS2 and UK 2030 + LS3 scenarios compared with the UK 2030 + LS1 scenario, concentrated in London. In UK 2030 + LS1. there are also health benefits from other health outcomes (average cases per year) related to both respiratory (e.g. 388,000 asthmatic symptom days in asthmatic children) and cardiovascular diseases (e.g. 3077 new cases of coronary heart disease). Again, there are further increases for UK2030+LS2, concentrated in London and the health benefits from the other health outcomes are marginally larger again for UK2030+LS3.

One obvious question is whether the estimates could possibly be accurate when predicting so far into the future. There is some truth in this, but we know for sure that the benefits will be underestimated if, for example, the analysis had only been done for the years 2018-2030. The air pollution reductions could have contributed to less initiation of disease and avoidance of mortality that would have occurred beyond 2030. ONS birth projections and mortality rate projections have been incorporated which, while also uncertain, covers at least one aspect of future trends. It is also likely that further policies for further reductions will be developed beyond 2030, at which point the analyses will be repeated. So the process is best seen as part of a package of continually updating analyses that predict into the future to the best of our ability with constant updates over time.

Some of the health outcomes quantified for this report have a long history of quantification e.g. mortality benefits and reductions in hospital admissions. Others are well established health outcomes e.g. respiratory symptoms, but less commonly quantified. This is partly because assumptions have to be made about baseline rates which are not routinely collected. Other areas of evidence have become established in recent years (e.g. incidence of coronary heart disease) but quantification methods are not fully developed. Further thinking is needed as to how to deal with diseases that are risk factors for each other e.g. coronary heart disease and stroke. And other evidence may become further established to allow inclusion in the future e.g. dementia.

None of the above uncertainties take away from the fact that air pollution reductions aimed at attaining the 2005 WHO guideline for $PM_{2.5}$ are likely to deliver substantial health benefits.

11. Monetary benefits

11.1. Monetised benefits (UK) life years gained

We have assessed the monetised benefits for life years gained in the UK in line with our methodology set out in Section 8, and the health assessment of life years saved in every year to 2134.

Table 44 Monetised benefits of life years gained

	Valuation, £
Monetised benefits (base case), £ PV	218,164,799,145
Annualised monetised benefits (base case), £ PV annualised	1,864,656,403
Monetised benefits (upper case), £ PV	300,047,860,191
Annualised monetised benefits (upper case), £ PV annualised	2,406,204,202

Table 19 shows the economic benefits from air pollution reduction for Scenario UK2030+LS1. For the UK, as a whole, the monetised value of economic benefits from reduced air pollution life years saved is **£218 billion**. However, using a different assumption on the value of life, also in line with government guidance, this could be as high as £300 billion. Our base case has been selected on the basis of government advice which was last reviewed thoroughly in 2007.

11.2. Monetised benefits (Other health outcomes)

Monetised benefits (Post neo-natal mortality)

Monetised benefits from post neo-natal mortality (1-12 months) under Scenarios UK2030+LS1, UK2030+LS2 and UK2030+LS3 in net present value terms are summarised in Table 45. Additional policies implemented in more ambitious London scenarios are estimated to have a marginal impact on total benefits. This is because they capture relatively small changes in policy (e.g. UK2030+LS3 models the impact of reductions in the remaining proportion of wood burning) that apply only to London.

Table 45 Monetised benefits of post neo-natal mortality under different scenarios (in £)

	UK2030+LS1 vs 2018	UK2030+LS2 vs 2018	UK2030+LS3 vs 2018	UK2030+LS 2 vs UK2030+LS 1	UK2030+LS3 vs UK2030+LS2
PV	4,343,415,198	4,432,834,141	4,446,027,100	89,418,943	13,192,959
PV (annualised)	37,123,207	37,887,471	38,000,232	764,264	112,760

Monetised benefits (morbidity benefits)

The benefits of reducing morbidity (ill health) are quantified in line with the methodology outlined in Section 8. The largest morbidity benefits in terms of numbers of cases come from asthmatic symptoms in asthmatic children and chronic phlegm in adults, followed by chronic bronchitic symptoms in asthmatic children and acute bronchitis in children. On the other hand, the monetary valuation of these health benefits leads to different results. In monetary terms, the diseases that have the largest impact are chronic bronchitis (chronic phlegm) and coronary heart disease that collectively account for around 95% of total morbidity benefits. This is based on studies that capture the value society place on suffering from different illnesses, thus incorporating severity of disease as well as numbers of cases.

Table 46 Monetised benefits of morbidity under different scenarios (in £)

Morbidity	UK2030+LS1	UK2030+LS2	UK2030+LS3	UK2030+LS1 and UK2030+LS2 difference	UK2030+LS 2 and UK2030+LS 3 difference[a]
Chronic Bronchitis (chronic phlegm)	84,912,136,639	86,613,589,660	86,870,412,758	1,701,453,022	256,823,098
Coronary Heart Disease	37,176,184,822	37,699,793,059	37,699,793,059	523,608,237	-
Acute Bronchitis in children	2,550,757,520	2,603,275,257	2,610,759,006	52,517,737	7,483,748
Asthmatic symptom days	1,832,703,066	1,870,464,553	1,875,708,568	37,761,487	5,244,015
Bronchitic symptoms in asthmatic children (NO_2)	163,798,153	164,791,914	164,791,914	993,760	-
Hospital Admissions Cardiovascular (NO_2)	1,347,144,595	1,355,433,790	1,355,433,790	8,289,195	-
Hospital Admissions Respiratory (NO_2)	1,788,218,338	1,799,224,151	1,799,224,151	11,005,812	-
Subtotal	**129,770,943,133**	**132,106,572,384**	**132,376,123,245**	**2,335,629,251**	**269,550,861**

[a] The gaps in the far right column are for a couple of reasons (i) the changes between the two scenarios only related to $PM_{2.5}$ and some health outcomes are quantified on the basis of other pollutants (hospital admissions and chronic bronchitic symptoms in asthmatic children) or (ii) even if the health outcome is based on particulate matter, for rare outcomes the air pollution changes between the two scenarios are too small to save whole numbers of cases (coronary heart disease).

The method for hospital admissions calculates the results for both NO_2 and $PM_{2.5}$ and takes the largest number (adding them together would be an over-estimate) (see section 7 Health Impact Assessment Methods). The larger number was for NO_2 for all scenarios compared with 2018 concentrations remaining unchanged. But for the difference between UK2030 plus LS2 and UK 2030 plus LS3, the results for $PM_{2.5}$ are marginally larger because there is no change in NO_2 concentrations. The results for $PM_{2.5}$ and hospital admissions are £1.87 million and £26,310 for cardiovascular and respiratory diseases respectively.

11.3. Monetised benefits (healthcare sector costs)

Air pollution damages health by promoting the onset of some non-communicable diseases that can increase the cost to the healthcare system. Over the appraisal period (2020-2134), chronic bronchitis and coronary heart disease are collectively estimated to increase healthcare sector costs by £4.4 billion. Therefore, a reduction in air pollution related illnesses can deliver substantial savings to the healthcare system.

Table 47 Monetised benefits of healthcare sector costs (in £)

Health Sector Costs	Scenario UK2030+LS1	Scenario UK2030+LS2	Scenario UK2030+LS3	Scenario UK2030+LS1 and UK2030+LS2 difference	Scenario UK2030+LS2 and UK2030+LS3 difference
Coronary Heart Disease	2,028,150,950	2,056,716,456	2,056,716,456	28,565,506	
Chronic Bronchitis (chronic phlegm)	2,373,453,729	2,421,012,537	2,428,191,225	47,558,808	7,178,688
Subtotal	**4,401,604,679**	**4,477,728,994**	**4,484,907,682**	**76,124,315**	**7,178,688**

11.4. Monetised benefits labour market impacts

People taking time off work due to air pollution related illnesses, or to care for dependents that are ill, costs the UK economy £18 billion over the period to 2134. In monetary terms, chronic bronchitis has the largest impact on workplace absences. In addition, workers may turn up to work but not be as productive because they are ill. This costs the UK an additional £9.5 billion over the period to 2134. Therefore, reducing air pollution related illnesses, particularly chronic bronchitis, can deliver economic gains.

Table 48 Monetised benefits of absenteeism (in £)

Health Sector Costs	**UK2030+LS1**	**UK2030+LS2**	**UK2030+LS3**	**UK2030+LS1 and UK2030+LS2 difference**	**UK2030+LS2 and UK2030+LS3 difference**
Chronic Bronchitis	15,714,213,640	16,029,092,023	16,076,620,835	314,878,383	47,528,813
Coronary Heart Disease	110,462,643	112,018,455	112,018,4h55	1,555,812	-
Child health related absenteeism	2,199,428,341	2,243,395,176	2,156,697,221	43,966,835	86,697,955
Subtotal	**18,024,104,624**	**18,384,505,654**	**18,345,336,512**	**360,401,030**	**39,169,142**

Table 49: Monetised benefits of presenteeism under different scenarios (in £)

Health Sector Costs	**UK2030+LS1**	**UK2030+LS2**	**UK2030+LS3**	**UK2030+LS1 and UK2030+LS2 difference**	**UK2030+LS2 and UK2030+LS3 difference**
Chronic Bronchitis	£9,428,528,184	£9,617,455,214	£9,645,972,501	188,927,030	28,517,288
Coronary Heart Disease	£24,726,690	£25,074,953	£25,074,953	348,263	-
Subtotal	£9,453,254,874	£9,642,530,167	£9,671,047,455	189,275,293	28,517,288

12. Assessment of existing policies

The modelling results described in this report have shown that poor air quality is linked to adverse health outcomes. There are a wide range of policy options that can improve air quality at both a UK-level and for cities that have higher levels of air pollution. The objective of this work is to understand the rationale for policies affecting air pollution and compare different policies using economic appraisal tools. This can identify the most cost-effective interventions that will deliver socio-economic benefits.

Our analysis (Section 11) indicates that a reduction in air pollution following existing government policies and government net zero commitments could lead to total benefits of £384 billion – these benefits justify annual expenditure on both new and existing policies of up to £3.3 billion to 2134. This section reviews the costs and benefits of key policies that have already been implemented by government as part of the UK2030 scenario. Our findings in this section demonstrates that:

1) Policies included in our analysis can be justified on a standalone cost benefit analysis (using government data sources for both costs and benefits).
2) The majority of air pollution benefits arise as co-benefits alongside energy and carbon savings – maximising these benefits requires continued commitment but not necessarily large additional expenditure.

Section 8 set out the causal pathways showing how reductions in air pollution can improve socio-economic outcomes. This section is structured as follows:

- Section 12.1: Sets out the methodology used to appraise the costs and benefits of key air pollution policies using government impact assessments.
- Section 12.5: Sets out the findings of the cost-benefit assessment, providing further details on key air pollution policies that deliver cost-effective improvements in health and economic outcomes.

12.1. Methodology to appraise costs and benefits

Figure 29 sets out the approach adopted to appraise the costs and benefits of policies included in the UK2030 scenario (excluding London policies). This analysis draws upon UK Government impact assessments with a focus on policies that are likely to have a material impact on air pollution[104]. Policies included in the analysis are expected to deliver emission

[104] This analysis is based on an ex-ante assessment of costs and benefits. In practice, the costs and benefits could diverge from current estimates. For example, a study on the regulation of industrial water pollution in the US found that capital costs were overestimated by 72% and operation and maintenance costs by 117%. A report by the California Air Resources Board on the cost of adopting regulations to reduce GHG emissions from vehicles, found that regulators overestimated the costs by between 20-80% (Defra, 2007). Equally, it is also possible that ex-ante costs are underestimated and end up imposing a higher burden on consumers (businesses and households).

reductions over the period 2018-2030 (with the benefits aggregating over a longer time period).

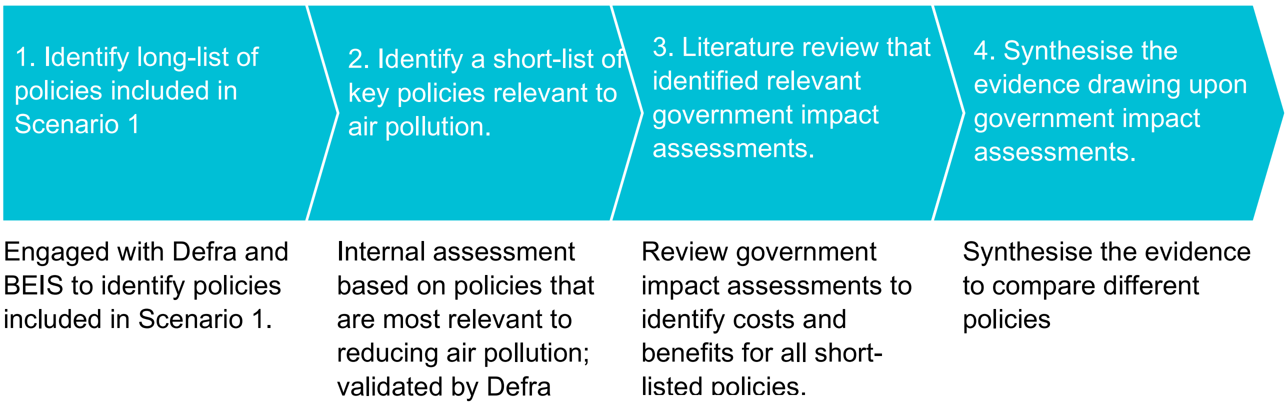


Figure 29 Process of reviewing a shortlist of government policies

Scenarios LS2 and LS3 model more ambitious reductions in air pollution in London. Unfortunately, there is currently insufficient clarity around the policy details to estimate the costs of polices being considered by the London Government.

12.2. Long-list of baseline policies

Defra's baseline projections used in the UK2030 scenario are largely based on policies outlined in BEIS' 2018 Energy and Emission Projections (EEP)[105]. The EEP projects future energy use and greenhouse gas emissions in the UK considering around 65 climate change policies where funding has been agreed, or policy design is sufficiently advanced to allow robust estimates of policy impacts to be made.

Given the nature of the EEP, several of these policies were introduced to reduce GHG emissions, and they do not directly map across to policies that are relevant from an air pollution perspective[106]. However, many climate change and energy policies typically deliver co-benefits in terms of improved air quality. For example, a switch from coal fired plants to natural gas in the 1990s played an important role in reducing air pollution, particularly SOx and particulate matter, although these were not purely motivated by air pollution considerations. Future climate policies are also likely to deliver co-benefits. A ban on internal combustion vehicles and a transition to low-carbon vehicles to achieve net-zero emissions will produce substantive benefits in terms of air pollution reduction, particularly NOx, and $PM_{2.5}$. In addition to EEP policies, the baseline includes policies introduced by Defra to directly reduce air pollution, in particular regulations covering biomass burning.

[105] Policies covering biomass are included in the baseline but are not part of the EEP.

[106] Air pollution includes pollutants (NO_X, SO_2, $PM_{2.5}$, PM_{10}) that have a material impact on human health.

12.3. Short-listed policies

We conducted an internal assessment to identify a short-list of EEP policies that are likely to have a material impact on air pollution. This was based on policies that are likely to deliver significant reductions in air pollution, and the feasibility of finding government information on costs and benefits[107]. The list was refined following conversations with relevant government departments. The policies that we short-listed are covered in Table 50.

Table 50 *Short list of policies most relevant to air pollution reduction*

Sector	Key Policies	
Energy and Industry	Industrial Emissions Directive (replaced the Large Combustion Plant Directive)	Reduce and restrict air pollution emissions from large combustion units (>50MWt)
	Medium Plan Combustion Directive	Reduce and restrict air pollution emissions from combustions units below the IED size of >1MWt
Transport	Balanced Net Zero Pathway for Transport (CCC's net-zero pathway replaces transport sector policies in Defra's baseline)	UK government advisory on projected transport pathway to achieve net zero targets by 2050, including assumptions on electric vehicle uptake and phase out of conventional fuels.
	Euro 6/VI Standards	EU rules on minimum air pollution standards for diesel vehicles
Biomass	Regulations covering wood burning and coal	Restricting the sale of less energy dense wood and coal fuel products. Reducing air pollution caused by the combustion of both fuels
Buildings	Building regulations 2010	Additional measures to improve heat insulation
	Building regulations 2013	Additional measures to improve heat insulation
	Technical standards for boilers (Boiler Plus)	Policies to improve gas boiler efficiency.
	Private Rented Sector Energy Efficiency Regulations	Energy efficiency measures primarily through improved insulation
	Heat network investment project (Green Heat Network Fund)	Grant scheme supporting development of low carbon heat networks - likely to reduce fuel combustion and air pollution.

[107] We do not include expired policies in our assessment, although some of these policies could continue to deliver some reductions in air pollution.

There are some EEP policies that are likely to have a positive impact on air pollution but have been excluded from this study because it is challenging to accurately estimate costs and measure impact. For example, policies that encourage fuel switching (e.g. to renewables) and investment in energy saving technologies are likely to have a positive impact on air pollution. There are a combination of UK and EU policies that strengthen the business environment to invest in such measures (e.g. EU ETS, Contracts for Difference, carbon price floor). However, it is difficult to robustly measure the impact on reducing air pollution, and the associated costs of these policies because they depend on which technological options firms decide to adopt. For example, the EU ETS covers electricity and heat generation, and energy-intensive industries. Firms have a range of options to reduce emissions, and the impact on air pollution will depend on the technology adopted e.g. whether a particular plant switched from coal to natural gas, renewables, or biomass.

12.4. Approach

Our approach to reviewing different policy impact assessments consistently involves:

- **Annualising costs and benefits:** We estimated the annualised costs and benefits of a policy by calculating the net present value (NPV) and dividing the NPV over the lifetime of the policy.
- **Ensuring a consistent time period:** This study covers the period 2018-2030. However, the policies in the baseline have different timeframes. Many policies were introduced before 2018 but the benefits are likely to continue to be realised over the period 2018-2030. For some polices, the benefits are likely to extend beyond 2030. To ensure policies can be easily compared, we adjust the annualised costs and benefits to focus on the period 2018-2030.
- **Keeping a consistent price base:** This is done by rebasing all costs and benefits 2018. The aim of our comparisons is to show the scale of real annualised costs and benefits between 2018 – 2030 for short-listed policies.

12.5. Findings of the cost-benefit assessment for the UK2030 scenario

- Air pollution policies can be justified on a standalone basis with the benefits outweighing the costs of implementation.
- Policies related to industrial emissions (Industrial Emission Directive and the Medium Combustion Plant Directive), transport (switch to low-carbon transport in line with the CCC's net-zero scenario) and wood burning (regulations covering biomass burning and coal) deliver large air pollution benefits in a cost-effective manner.
- Several other policies deliver reductions in air pollution as a co-benefit. For these policies, the air pollution benefits are relatively smaller compared to benefits produced from energy savings and a reduction in GHG emissions. This includes regulations covering the buildings sectors, and interventions to encourage a switch to cleaner heating fuels.

The results from the cost-benefit assessment, drawn from government impact assessments are shown in Table 51.

Table 51 Benefit cost ratios and economic appraisal of shortlisted policies

Sector	Policy	PV £ million 2018[108]		Benefit cost ratio
		Benefits	Costs	
Energy and Industry	Industrial Emissions Directive (Upper and Lower Scenario)	6,748 -10,650	2,927- 1758	2.3 – 6.1
	Medium Plan Combustion Directive	1,082	224	4.8
Transport	Balanced Net Zero Pathway for Transport (replaces transport sector policies in Defra's baseline)*	690,558	182,500	3.8
	Euro 6/VI Standards			NA
Biomass	Regulations covering wood burning and coal	8,141	148	55
Buildings	Building regulations 2010	45,924	23,126	2.0
	Building regulations 2013	1,669	1,245	1.3
	Technical standards for boilers (Boiler Plus)	1,526	1,025	1.5
	Private Rented Sector Energy Efficiency Regulations	1,517	926	1.6
	Heat network investment project (Green Heat Network Fund)	1,179	589	2.0

[108] **IED** Updated Impact Assessment of the Industrial Emissions Directive (IED): Large Combustion Plants produced by Amec for Defra 2011 / Updated Impact Assessment of the Industrial Emissions Directive (IED): Large Combustion Plants produced by Amec for Defra 2012. **MPCD** Amendments to environmental permitting regulations to improve Air quality by transposition of the Medium Combustion Plant Directive , BEIS/Welsh Government, 2017. **Balanced Net Zero Pathway** CCC surface transport sector Sixth Carbon Budget, CCC. **Euro 6/VI Standards** Consultant estimate (explained in the Technical note). **Regulations covering wood burning and coal** Proposed regulation of the sales, distribution and marketing of: Wet wood (>20% moisture) sold in units up to 2m3; Bituminous house coal; Banning manufactured solid fuels with sulphur content over 2% Defra 2019. **Building regulations 2010** Implementation Stage Impact Assessment of Revisions to Parts F and L of the Building Regulations from 2010, Department for Communities and Local Government (2010). **Building regulations 2010** Changes to Part L of the Building Regulations 2013, Department for Communities and Local Government 2013. **Technical standards for boilers** Domestic Heating Replacement Regulations, BEIS 2017. **Private rented-sector energy-efficiency regulations** Final Stage Impact Assessment: Amending the Private Rented Sector Energy Efficiency Regulations, BEIS 2018. **Green Heat Network Fund (GHNF) Consultation IA** The heat network investment project (Green Heat Network Fund), BEIS 2020

Below we describe in further detail the costs and benefits of three policies that deliver significant reductions in air pollutants. This includes the Industrial Emissions Directive (IED), the Medium Combustion Plant Directive (MCPD) and regulations covering domestic biomass burning.

Industrial Emissions Directive

The Large Combustion Plant Directive, in place since 2007 was replaced by the IED which was expected to be implemented in 2016. Both directives were introduced to improve air quality. Combustion plants greater than 50 MW have a range of options to comply with new air pollution standards. The government estimated the costs and benefits for two scenarios that reflect the different options available to large combustion plants. The benefits outweigh the costs under both scenarios.

The European Directive allowed the government to develop a Transitional National Plan (TNP) to implement the IED recognising the significant costs and change in behaviours required to achieve compliance. In 2012, the government intended to implement IED emission limit values by 2016. However, noting the scale of challenge, the government launched a consultation in 2015 to develop a TNP that gradually decreases plant emission limits to June 2020, by which point the IED emission limit values would be fully enforced. The delay in fully implement IED emission limits resulted in additional financial savings for large combustion plant operators, and a reduction in air pollution benefits. The government argued that not implementing the TNP would have resulted in several plants coming offline, creating potential risks for energy security and resilience.

Following the introduction of the TNP, large combustion plant operators, typically part of the electricity supply industry have additional time to invest and install abatement technologies. Other sectors that have benefited from the TNP include oil refineries, iron and steel companies and other energy-intensive industrial sectors. However, since the TNP is expected to end in 2020, the UK is expected to see improvements in air pollution as larger plants comply with limits set out in the IED.

The IED imposes additional costs on large industrial plants. Industrial plants have two options:

- Plants can either close operations before the end of their useful life because they are unable to invest in abatement technology to comply with emission limits. This will result in a loss in profit for these operators, and potential job losses at these plants.
- In order to stay open, plants need to invest in abatement technologies to reduce their NOx, SO_2 and PM emissions within the Emission Limit Values set out by the regulator. These one-off transitional costs vary from £1.2 billion to £ 1.5 billion to achieve compliance (2018 values).

In addition, the policy marginally increases administrative costs faced by the regulator and operators. This includes the administrative costs operators face to apply for environmental permits, and the cost to regulators to process applications (which are passed on to operators). Administrative costs are expected to be around £3,300 per plant affected by the IED.

The benefits delivered by IED are expected to outweigh the costs because of the significant air pollution benefits that can be realised once large combustion plants begin to meet the emission limits from 2020 onwards.

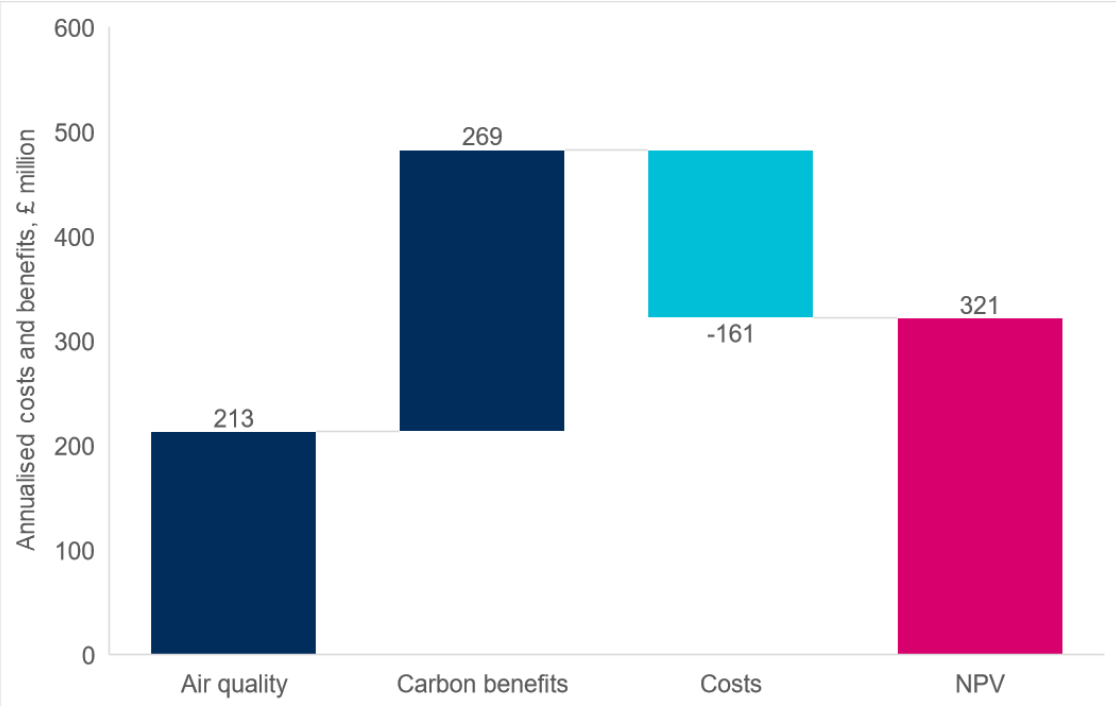


Figure 30 Industrial Emissions Directive (Upper Scenario), Annualised costs and benefits 2018 - 2030

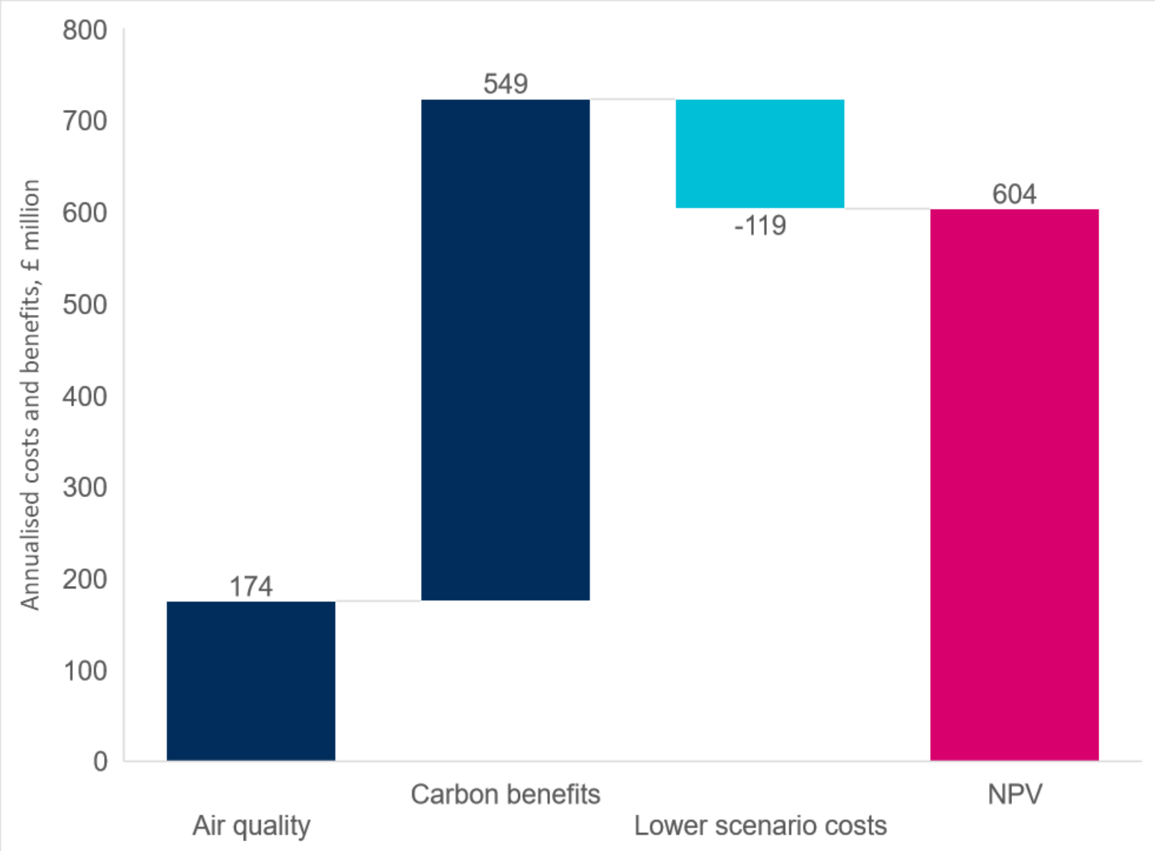


Figure 31 Industrial Emissions Directive (Lower Scenario), Annualised costs and benefits 2018 - 2030

Medium Combustion Plant Directive

The MCPD was introduced in 2018 with the primary aim of reducing air pollution from combustion plants between 1 MW and 50 MW. Before the introduction of the MPCD, emissions medium combustion plants were largely unregulated in the UK (Defra, 2017[109]).

Similar to the IED, the main costs of the MCPD are borne by plant operators that will need to install abatement technologies to meet stricter emission limits. The annual costs to plant operators between 2018-2030 in the range of £72 million - 278 million (2018 prices). In addition, there will be administrative and monitoring costs in the range of £3 million - £10 million (2018 prices).

Most MCPD plants typically install NOx abatement technology (e.g. low NOx burners) to achieve emission targets, which delivers significant air pollution benefits but only marginal GHG reduction benefits, as shown in Figure 32. The benefits of clean air are borne by society as a whole, particularly those people that live and work near combustion plants.

[109]Defra Impact Assessment – Amendments to environmental permitting regulations to improve air quality by transposition of the Medium Combustion Plant Directive.

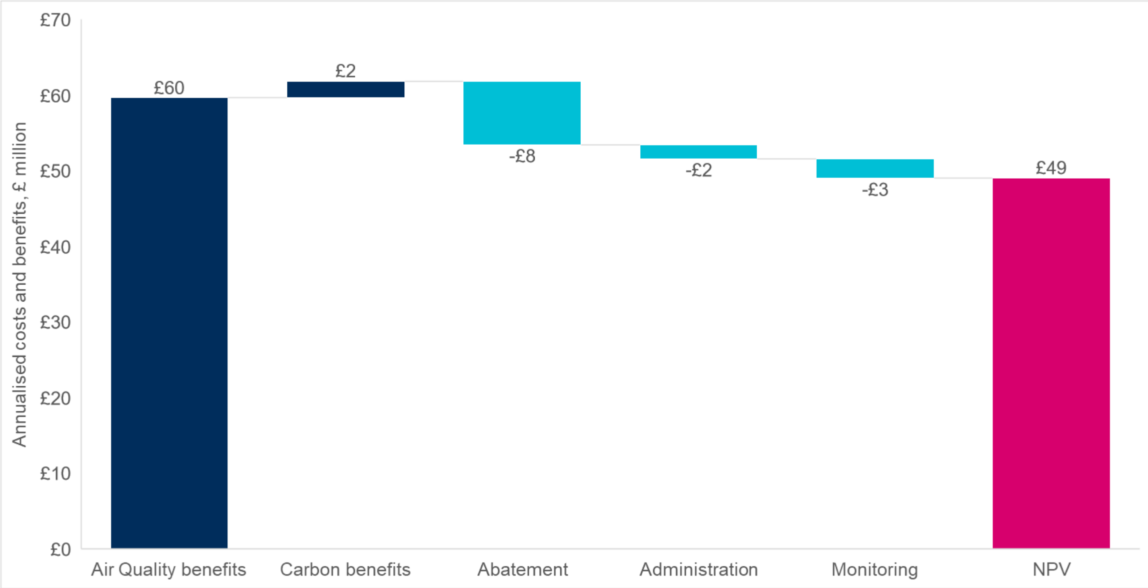


Figure 32 Medium Plant Combustion Directive, Annualised costs and benefits 2018 - 2030

Regulations covering wood burning and coal

Domestic burning of solid fuels is by far the largest source of $PM_{2.5}$ emissions (Defra, 2019[110]). This imposes costs on society in the form of ill health. However, these costs are not fully considered by the domestic fuel market. In 2019, Defra consulted on restrictions covering domestic fuels that are more polluting and have a lower energy content compared to alternatives. Restricting the use of these fuels, namely wet wood, bituminous house coal and other manufactured fuels with high sulphur content will significantly reduce air pollution at relatively low cost.

Restricting the use of these fuels impose additional costs on businesses and households.

- **Businesses:** Businesses are likely to experience increased costs, including:
    - **Administrative and monitoring costs:** Administrative costs are costs borne by the fuel manufacturer as part of the inspection. This typically includes the costs time spent by the manufacturer's quality control manager with the regulatory agency assessing fuel production and quality control records. Monitoring costs are enforcement costs borne by the regulator including the cost of regular inspections and conducting tests on fuels sold in the market. These are passed on to fuel manufactures in the form of fuel testing charges and annual registration fees. Administrative and monitoring costs are likely to range between £22 million - £33 million in 2018 present value terms.
    - **Capital costs:** in order to comply with new regulations, businesses will need to invest in drying facilities (e.g. drying kiln or a covered space to season wood). This is likely to cost businesses £70,000 - £80,000 for a drying capacity of 1,800

[110] Defra Impact Assessment – Proposed regulation of the sale, distribution, and marketing of wet wood, bituminous coal, and banning manufactured solid fuels with Sulphur content over 2%.

tonnes of wet wood per year, with total costs to industry in the range of £68 - £101 million in 2018 values.

  - **Operational costs:** Businesses are also likely to faced increased operating costs, including labour, maintenance, insurance, and other feedstock costs.
  - **Loss in profit:** The regulation is designed to encourage a shift to higher density, more efficient fuels. This will result in a reduction in the volume of fuel sold (because the same volume of fuel can generate more energy), which will translate into lower industry profits. The government estimates the loss of profit to be around £15 million in present value terms. This accounts for some of the profit loss being offset by an increase in the sale of alternative fuels.

- **Households:** Households are likely to face higher upfront costs as they shift from burning wet wood and traditional bituminous coal to dry wood and low-sulphur manufactured fuels. However, for most households, these high upfront costs are likely to be offset by energy savings driven by a switch to fuels with higher energy density. According to analysis commissioned by Defra, dry wood can be 17% cheaper on an energy adjusted basis compared to wet wood, and manufactured solid fuels are 6% cheaper compared to traditional coal. Households that currently burn high sulphur manufactured solid fuel are likely to face higher costs. This is expected to affect 12,500 households, who on average are likely to experience an increase in costs of £170 per year between 2020-2030.
- **Government:** There are marginal costs to government (approx. £220,000 over three years) from running information campaigns to promote safer and cleaner fuels. In addition, the enforcement costs are likely to be in the range of £1.2 million over the 11 year period.

The policy is expected to deliver substantial environmental benefits that offset the increased costs to business. Many of the costs are offset by energy savings that households are expected to experience as they use more energy efficient fuels.

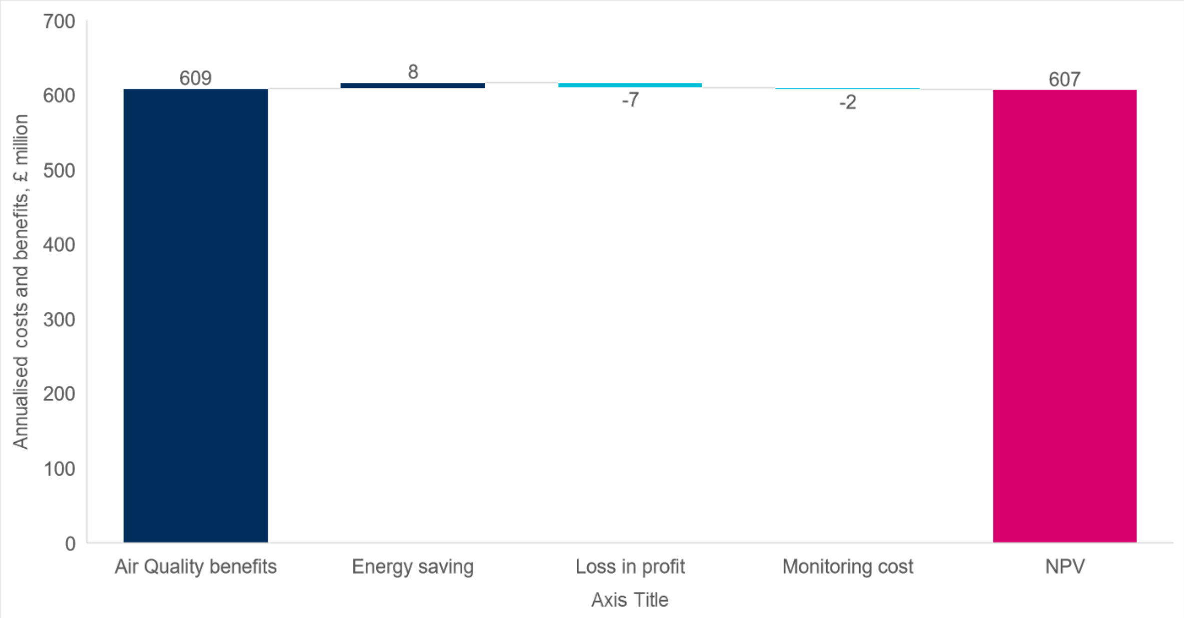


Figure 33 Regulation of the sales, distribution and marketing of home burning fuels 2018 - 2030

12.6. Transport Policy

The modelling carried out for this study replaces the current Defra baseline policies with CCC's Balanced Net Zero Pathway (section 3.2). The CCC pathway includes an accelerated shift to low-carbon vehicles to ensure the UK is on track to achieve net-zero emissions by 2050.

The CCC estimated the capital and operational costs of a shift to low-carbon vehicles (e.g. the cost of installing a network of electric chargers, and operational costs/savings to consumers from switching to low-carbon vehicles). This has been complemented by estimating the carbon benefits of the policy using BEIS central estimate for policy appraisal carbon price over the period 2018-2030[111] – this sets out a price of £241/tonne in 2020 and £280/tonne in 2030.

The policy has a positive benefit cost ratio of 3.8 without accounting for additional air pollution benefits. The CCC estimates that the operational savings will be sufficient to cover the additional capital expenditure (e.g. electric charging infrastructure) required to fund the transition to net-zero.

[111] https://www.gov.uk/government/publications/valuing-greenhouse-gas-emissions-in-policy-appraisal/valuation-of-greenhouse-gas-emissions-for-policy-appraisal-and-evaluation (Accessed 09 February 2022).

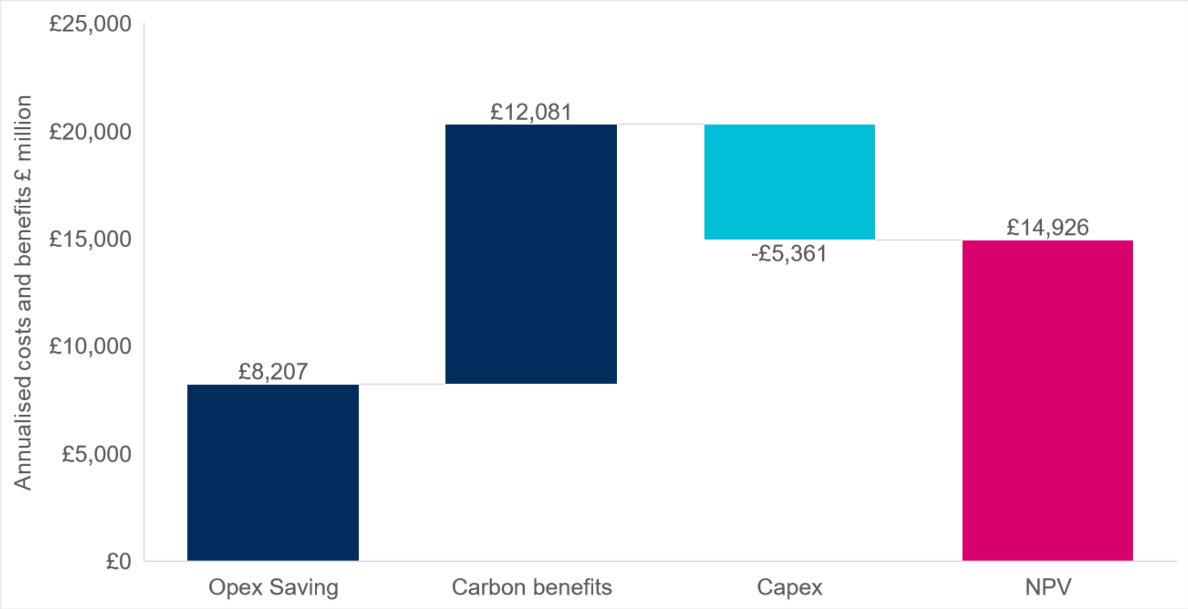


Figure 34 CCC Balanced Pathway Surface Transport, Annualised costs and benefits 2018 - 2030

The CCC analysis does not quantify air quality improvements as a benefit, which will increase the benefits of the policy. It has not been possible to attribute reductions to air pollution to specific policies. The health and economic benefits of a shift to low-carbon vehicles is included as part of our overall benefits assessment in section 10 and 11.

In addition, tightening existing vehicle pollutant emission standards will reduce emissions from new petrol and diesel vehicles. This will impose additional costs on manufacturers, which may be passed on, in part or in full, to consumers in the form of higher prices. Our estimation of Euro 6/VI costs considers:

- The CCC budget's expected vehicle km for diesel vehicles
- The expected lifetime of diesel vehicles by road km[112]
- European estimates of additional cost of a Euro 6/VI vehicles (€ 275) [113] [114]

The compliance costs of achieving Euro 6/VI standards are £134 million per annum in 2018 prices.

12.7. Other policies

There are other policies included in the baseline that are also expected to deliver reductions in air pollution. In the buildings sector, regulations typically incentivise developers and

[112] https://www.transportenvironment.org/wp-content/uploads/2021/07/2017_09_Diesel_report_final.pdf (Accessed 09 February 2022).

[113] https://ec.europa.eu/commission/presscorner/detail/en/MEMO_06_409 (Accessed 09 February 2022).

[114] The estimate is converted into GBP using the exchange rate 1 Euro= 0.8599 GBP

owners to improve energy efficiency and adopt higher heating standards (in both new and existing buildings). Recent policies, such as the development of green heat networks have focused on carbon reduction rather than air pollution. The air pollution benefits from these policies covering the building sector range from 0.5% to 7% of total benefits with the exception of private sector rental regulations, where air pollution benefits account for 17% of total benefits. Air pollution benefits from private sector rental regulation arise from decreased heating requirements and use of alternative non-polluting generation sources e.g. solar thermal.

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