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Air Quality Damage Cost Update 2025

Report for Defra

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Executive summary

Air pollution can have damaging impacts on human health, productivity, amenity and the health of the environment. These detrimental impacts have associated socio-economic costs that are not captured in the market price of the goods or services consumed that produce the pollution (also known as external costs or externalities). The UK Department of Environment, Food and Rural Affairs (Defra) has produced guidance (Defra, 2025) to steer the assessment of air quality impacts and the valuation of external costs such that these can be captured in policy appraisal.

Defra's guidance provides approaches for assessing and valuing the impacts, one of which is the 'damage cost' approach. The guidance provides recommendations for when to apply this simplified approach. A set of 'damage cost' values estimated per tonne of emissions of an air pollutant have been calculated. These values capture the external costs associated with a marginal change in air pollutant emissions. They can be combined with forecasts of changes in air pollutant emission to provide an approximate valuation of the impacts of a policy.

Ricardo has been commissioned by Defra to update the damage costs of air pollution. Seven technical tasks were included in this update, comprising:

1. Updating the emissions-to-concentration modelling for NOx $PM_{2.5}$, SO_2 and NH_3 to 2023.
2. Updating the emissions-to-concentration modelling for O_3 precursors to 2022.
3. Aligning discounting and value of one Quality-Adjusted-Life Year (QALY) with the HMT Green Book.
4. Adding new domestic damage costs split by size of urban area (area type).
5. Updating all underlying population and baseline health data with evidence published.
6. Including a breakdown of damage cost values and health impact metrics (e.g., QALYs) per tonnes of emissions by specific health pathways, where available.
7. Conducing additional sensitivity tests, in addition to low, central and high damage cost estimates.

The updated set of damage costs is presented in Table 0 below, alongside the low and high estimated sensitivities around the central values. This table shows the national average damage cost values per tonne of air pollutant emissions, by selected pollutant. Sector-specific damage costs have also been updated and are also presented in this report. A positive damage cost value means that an increase in air pollutant emissions generates a cost to society, and a decrease in emissions generates a benefit.

Table 0-1-1 Revised national average damage cost estimates and sensitivity bounds (2025 prices, 2025 impact year). $PM_{2.5}$ is the preferred metric for the change in PM emissions

Pollutant Emitted	Damage Cost – Central estimate (£/tonne)	Damage Cost – Low estimate (£/tonne)	Damage Cost – High estimate (£/tonne)
NOx	10,193	1,671	34,546
SO_2	26,193	9,158	72,022
NH_3	9,900	3,376	27,868
VOC	150	90	277
$PM_{2.5}$	114,411	40,026	371,654

Splitting the results by their contributing pathways, the effects of long-term exposure on mortality are the dominant impact captured in these damage cost estimates. This effect is captured alongside the effects of air pollution on hospital admissions (associated with short-term exposure), ischaemic heart disease, stroke, lung cancer, asthma in children, economic productivity, ecosystems, material damage and building soiling in the revised costs. The high estimate of damage costs includes additional impacts on morbidity, such as chronic bronchitis, for which there is greater uncertainty.

Each of the updated damage costs shows some variation relative to the set of damage costs published in 2023 (Ricardo, 2023). The changes observed are the result of the various changes made in the estimation of the damage costs, some of which have an inflationary and some a deflationary effect. There was a relatively high level of inflation between 2022 and 2025 of 15%. There were also increases in UK population of about 2% and baseline death rate of about 7%. This leads to a systematic increase in the contribution of the most important mortality and morbidity pathways to the overall damage costs. In more detail:

- The 2025 damage costs for NOx have a similar magnitude to the 2023 damage costs (when expressed in 2025 prices), once the changes in death rates, population and inflation are accounted for.
- The 2025 damage cost for SO_2 are higher than the 2023 damage costs (when expressed in 2025 prices). This results from an increase in the μgm^{-3} of secondary $PM_{2.5}$ per tonne of SO_2 emitted, associated with a decline in SO_2 emissions but an increase in measured SO_2 particulate matter concentrations. Increases in population and death rates have contributed to this increase.
- The 2025 damage costs for NH_3 are lower than the 2023 update (when expressed in 2025 prices). This change is due to a reduction in the μgm^{-3} of secondary $PM_{2.5}$ per tonne of NH_3 emitted, which results from a decrease in measured NH_4 particulate matter concentrations, partly offset by increases in population and death rates.
- The 2025 damage costs for VOC[1] are also lower. The relationships between VOC emissions and O_3 concentrations have been recalculated for the 2025 damage costs using the chemistry transport model CMAQ. The recalculation reflects a combination of changes in atmospheric composition between the previous modelling year of 2014, the new modelling year of 2022 and differences in model formulations and assumptions.
- The 2025 damage costs for $PM_{2.5}$ increased relative to the 2023 damage costs (expressed in 2025 prices). This increase is the result of an increase in the μgm^{-3} change in primary $PM_{2.5}$ concentrations per tonne of $PM_{2.5}$ emission. This increase is primarily associated with a reduction in the estimates of UK $PM_{2.5}$ emissions between the 2019 NAEI that was used for the 2023 damage costs and the 2022 NAEI that was used for the 2025 damage costs, reflecting changes in assumptions regarding activity levels, emission factors and the mapping of emissions.

The damage costs have been revised to reflect specific improvements in the underlying evidence base. The appraisal guidance (Defra, 2025) provides recommendations for when to apply the damage costs. This is to reflect the implicit assumptions made when applying the damage costs: in particular, that patterns of pollutant emission and exposure and baseline population and rates of health incidence could change over time and inherently represent an averaging of effects across the country as a whole or specific sector defined by the damage cost applied.

Further, users of the damage costs should note wider caveats, in particular regarding the uncertainty associated with their estimation and the coverage of impacts included. Users are encouraged to refer to the wider Defra guidance and original damage cost documentation (AEA Technology, 2006) for further information. In particular, the damage costs only capture a sample of the range of impacts associated with air pollution, and some remain unaccounted for in the damage costs, including:

- The damage costs only account for impacts of UK emissions on the UK and not on other countries.
- Not all of the impact pathways included in PHE's 'Estimation of costs to the NHS and social care due to the health impacts of air pollution' (PHE, 2018) have been included. In addition, the epidemiological evidence base linking air pollutant exposure to a wide range of conditions continues to grow – only a selection of pathways for which more robust CRFs are available are captured here.

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[1] Throughout this report VOC means Non-Methane Volatile Organic Compounds.

- Some ecosystem impact pathways have been included based on the work of (Jones, Mills, & Milne, 2014) – those ranked as ‘robust’. However, other ecosystem service impacts have not been included.
- The damage costs for VOC include impacts via the O_3 pathways only.

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1 Introduction

1.1 Air quality and impact valuation

The quality of the air around us has a strong influence on both natural and man-made environments. Air pollution can have damaging impacts on human health, productivity, amenity and the health of the environment. These detrimental impacts have an associated economic or social cost (known as external costs or externalities) that are not captured in the market price of the goods or services consumed that produce the air pollution.

Cost-benefit analysis (CBA) is a tool commonly used to appraise options in Impact Assessments (IA) to support policy development. CBA attempts to value all the costs and benefits associated with a given policy option, including any external costs that are not captured by market prices. The UK Department of Environment, Food and Rural Affairs (Defra) has produced guidance (Defra, 2025) to steer the assessment of air quality impacts and the valuation of associated external economic and social costs, based on the work of the Defra-led Interdepartmental Group on Costs and Benefits (IGCB). This guidance supplements the Green Book (HMT, 2024) which provides wider guidance for IA and valuation. These processes are designed to support evidence gathering to inform policy development and evaluation.

Defra's air quality appraisal guidance provides approaches for assessing and valuing the impacts, one of which is the 'damage cost' approach. The guidance provides recommendations for when to apply this simplified approach.

1.2 Damage costs of air pollution

Where possible, IGCB recommend that the Impact Pathway Approach (IPA) should be used to appraise the external impacts of policies, projects or programmes on air pollution. The IPA charts a logical progression from a change in pollutant emissions, through to monetised impact. This is a more detailed modelling approach, which utilises specific information regarding the policy and its impacts on air pollution to produce a more rigorous estimate of the likely effects. The approach was advanced through a series of EC DG Research projects known as (ExternE, 2005) and was also extensively used previously in analysis of impacts at the UK and EU level.

However, the IPA is relatively resource intensive and may not be a proportionate approach in all policy appraisals. This is particularly the case where air pollutant impacts are ancillary to the central effects of the policy. As such, Defra commissioned (AEA Technology, 2006) to develop a set of 'air pollution damage costs.

Damage costs are representative estimates of the external costs associated with a marginal change in pollutant emissions. The costs are expressed per tonne of pollutant emission. They can be readily combined with forecasted changes in emissions to provide an approximation of the aggregate external costs. Damage costs represent the impacts of an average unit of emission in the UK. As such they necessarily imply a simplified approach relative to undertaking an assessment using the full IPA. A more rigorous assessment using the IPA would take into account all specific information regarding the nature and location of the specific change in pollutant emission.

The initial set of damage costs for appraisal in the UK were estimated in 2006 by following the IPA for a range of impact pathways to capture the effects of an average emission in the UK. Since this initial set was produced, several updates have been made to the damage costs. For example, slight amendments to the methodology underpinning the estimation of the damage costs were subsequently noted in Defra's Air Quality Strategy (or AQS) (Defra, 2007), and a further updated set were published in 2011 (Defra, 2011b).

An update to the damage costs was published by Defra in 2015 (Defra, 2015c). The key element of this update was to reflect recent developments in the underlying evidence base concerning the effects of long-term exposure to concentrations of nitrogen dioxide (NO_2) on mortality.

Another update was published in 2019 (Birchby D. , Stedman, Whiting, & Vedrenne, 2019). This update refined the calculation of these effects based on 'Associations of long-term average concentrations of nitrogen dioxide with mortality' (COMEAP, 2018). This report included refined recommendations for quantifying mortality effects on the basis of long-term average concentrations of nitrogen dioxide (NO_2) from the UK Committee on the Medical Effects of Air Pollutants (COMEAP). The 2019 update was a major update and included the following:

- Splitting NOx damage costs by sector
- A split of the industry PM and NOx damage costs for 'Part A' installations
- An update of the estimation of mortality effects of long-term exposure to NO_2 to reflect updated COMEAP guidance
- Inclusion of new impact pathways: long-term exposure to PM_{10} on chronic bronchitis, impacts of exposure to ozone, impact pathways included in PHE's Estimation of costs to the NHS and social care due to the health impacts of air pollution (PHE, 2018) and impacts on productivity
- Updated baseline data for health impacts and population
- Estimation of impacts on air pollution on ecosystems.

A further update was published in 2020 (Birchby D. , Stedman, Stephenson, Wareham, & Williams, 2020). The key update for this publication was an update to the emissions to concentration modelling to a reference year of 2018 (using NAEI 2017) for NO_x and NO_2, $PM_{2.5}$ and PM_{10} and SO_2. A further revision to the damage costs was published in 2023 (Ricardo, 2023). This updated the emissions to concentration modelling to a reference year of 2020 (using NAEI 2019) for NO_x and NO_2, $PM_{2.5}$ and PM_{10} and SO_2. 2. The CRFs for several pathways, including that linking mortality to long-term exposure to $PM_{2.5}$, were also updated to reflect latest COMEAP guidance. Damage costs for rail were split into area types for the first time in this update.

Several versions of the damage costs are referred to in this report. For clarity:

- The first set of damage costs produced in 2006 are referred to as the 'original damage costs'.
- The 2015 set of damage costs published by Defra are referred to as the '2015 damage costs'.
- The 2019 set of damage costs published by Defra are referred to as the '2019 damage costs'.
- The 2020 set of damage costs published by Defra are referred to as the '2020 damage costs'.
- The 2023 set of damage costs published by Defra are referred to as the '2023 damage costs'.

Linked to the damage costs, BEIS include in their Supplementary Green Book guidance (BEIS, 2021b) a set of air pollution activity costs. These activity costs define the impact of exposure to air pollution per unit of energy consumed (per kWh), rather than per tonne as the damage costs are expressed. That said, the damage costs are used as a direct input in the calculation of the activity costs so the two are consistent.

1.3 Project objectives and approach

Ricardo Energy & Environment has been commissioned by Defra to update the damage costs of air pollution to reflect developments in the underlying evidence base. Prior to the 2025 update, Ricardo undertook a scoping study to inform the recommendations for amendments and updates within this iteration. The scoping report considered eleven proposed options for inclusion within this update, which were assessed through a combination of desk-based research and engagement with the Interdepartmental Group on Costs and Benefits (IGCB). The scoping exercise reviewed options to provide greater granularity within the sensitivity analysis, the calculation of detailed underlying metrics to support the assessment of health impacts, and update ozone modelling, all of which have been incorporated within the 2025 update. Other proposed options, such as the expansion of ecosystem services or the incorporation of transboundary effects will be revisited in future updates.

The scope of the 2025 damage cost update included the following key actions:

- Update to the emissions-to-concentrations modelling for all pollutants including ozone
- Splitting domestic damage cost by area type
- Checks to ensure the damage costs are consistent with the recent update to HMT's Green Book (in particular around discounting)
- Include values for specific health-based metrics per tonne of emissions in addition to economic valuations, where available.
- The results from further sensitivity tests, in addition to the results for the low, central and high damage cost estimates.
- Complementary update to the air pollution activity costs.

It is important to note that the damage costs, as per the previous update, still only capture a sample of the range of impacts associated with air pollution, and some remain unaccounted for in the damage costs, including:

- The geographic scope of the analysis has not been revised for the updated damage costs. The updated damage costs only account for impacts of UK emissions on the UK and not on other countries.
- Some of the impact pathways included in PHE's 'Estimation of costs to the NHS and social care due to the health impacts of air pollution' (PHE, 2018) remain included. However, some impact pathways identified in this work have not been included due to lower confidence around the supporting evidence base
- Some ecosystem impact pathways have been included based on the work of (Jones, Mills, & Milne, 2014) – those ranked as 'robust' and 'acceptable'. However, pathways assessed as 'improvements desirable' have not been included.
- The damage costs for VOC[2] include impacts via the O_3 pathways only.

This report is structured as follows:

- Section 2 sets the changes made to the air pollutant modelling underpinning the damage costs
- Section 3 provides details of the approaches adopted for the different human health impact pathways and their valuation
- Section 4 outlines the approach to the valuation of non-human health effects
- Section 5 describes the approach to estimating activity costs
- Section 6 outlines the methodology for deriving damage cost sensitivities
- Section 7 presents the final set of updated damage costs and underlying heath metrics and presents some comparisons with previous versions
- Section 8 concludes by presenting the updated activity costs.

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[2] Throughout this report VOC means Non-Methane Volatile Organic Compounds.

2 Updates to air pollutant dispersion modelling

2.1 Introduction

The emissions to concentrations air quality modelling has been updated for the revised damage cost calculations. The following models have been used and these models are discussed below, including references to full descriptions:

- Relationship between changes in primary $PM_{2.5}$ emissions and $PM_{2.5}$ ambient concentrations for total emissions and for individual emission sectors (Pollution Climate Mapping (PCM) model)
- Relationship between changes in NO_x emissions and NO_2 ambient concentrations for total emissions and for individual emission sectors (PCM model)
- Relationship between changes in SO_2 emissions and SO_2 ambient concentrations for total emissions (PCM model)
- Relationship between changes in SO_2, NO_x and NH_3 concentrations and ambient concentrations of secondary inorganic aerosol (a component of ambient PM_{10} and $PM_{2.5}$) (PCM model emission sensitivity coefficients method).

The following emissions to concentrations air quality modelling have also been updated

- Relationship between changes in NO_x emissions and ambient O_3 concentrations (Community Multiscale Air Quality Modelling System) (CMAQ) model)
- Relationship between changes in VOC emissions and ambient O_3 concentrations (CMAQ model).

The emissions to concentration modelling for $PM_{2.5}$, NO_2 and SO_2 was updated for the previous damage cost update that was published in 2023 based on the reference year of 2020 and using NAEI 2019 emissions. The O_3 modelling that was used for the damage cost update published in 2023 the same as was used for the damage costs published in 2019 for the reference year of 2014 and using NAEI 2013 emissions. We have previously recommended that the O_3 modelling be updated once every ~10 years and the new modelling incorporated into this current update is consistent with this recommendation. This recommendation was made based on a balance between the additional cost of undertaking modelling for ozone and the fact that atmospheric composition will be subject to gradual changes on the timescale of decades rather than years.

2.2 PCM model for the contribution of primary emissions to ambient concentrations

2.2.1 National damage costs

The PCM model has been used to calculate annual mean concentrations of PM_{10}, $PM_{2.5}$, NO_2 and SO_2 for 2023 on a 1 x 1 km grid. This model has been described in detail by (Pugsley, et al., 2025). 2023 is the most recent year for which PCM models are available for this update.

Overall, the modelling approach adopted is very similar to that used for the 2023 damage costs, which were based on air quality modelling for 2020. The changes in the emissions to concentrations relationships derived from the 2020 air quality modelling largely therefore reflect changes in the input datasets for the modelling relative to 2020. The inputs for the 2023 modelling include emission inventory estimates from the 2022 NAEI and ambient air quality measurement and meteorological data for 2023.

The PCM model results for each pollutant include contributions from a range of different sources. The calculation of damage costs requires the relationship between UK ambient concentrations and UK emissions (expressed as µgm^{-3} per tonne). Thus, only the sources within the PCM model that are related to UK emissions are relevant to the calculation of damage costs. These are the following contributions:

- Large point sources, modelled explicitly using the dispersion model ADMS
- Small point sources, modelled using a dispersion kernel approach (The model is run once for a unit emission rate from a single source and this is used to generate a dispersion kernel, which can be used to calculate concentrations from all sources considered).
- Area sources, modelled using the small points dispersion kernels for industrial emissions and dispersion kernels for other area sources, including kernels incorporating time varying emissions for domestic and road traffic sources.
- Regional concentrations of primary PM, modelled using the chemistry transport model TRACK.

The total concentrations of primary PM_{10}, $PM_{2.5}$, NO_2 and SO_2 associated with UK emissions inventory sources were calculated by summing these contributions and the population-weighted mean annual mean concentrations for 2023 were calculated for each pollutant using 1 x 1km population data from the 2021 census updated to be applicable for 2022. The µgm^{-3} per tonne for each pollutant was then calculated by dividing this population-weighted mean by the 2023 UK total emissions for each pollutant that were used to calculate the ambient concentrations within the model. The emissions for 2023 were calculated by scaling data from the NAEI for 2022 forwards by one year using emission projections provided by the NAEI as described by (Pugsley, et al., 2025).

The impact of primary emissions of NOx on concentrations of NO_2 is expressed as ugm^{-3} of NO_2 per tonne of NO_x emitted. This has been calculated by multiplying the ugm^{-3} of NO_x per tonne of NO_x emitted by the total UK population-weighted mean of NO_2 from all sources divided by the by the total UK population-weighted mean of NO_x from all sources.

2.2.2 Sector specific damage costs

The approach described above provides the average relationship between emissions and the exposure of the UK population to ambient concentrations. The impact of emissions on exposure to ambient concentrations varies for different sources and geographically, since it depends on the release characteristics of the emissions and the proximity of these emissions to centres of population. We have calculated emissions estimates for each sector and have run the concentration models on a sector by sector basis. We have used this to calculate the change in concentration per unit emissions for each emissions sector.

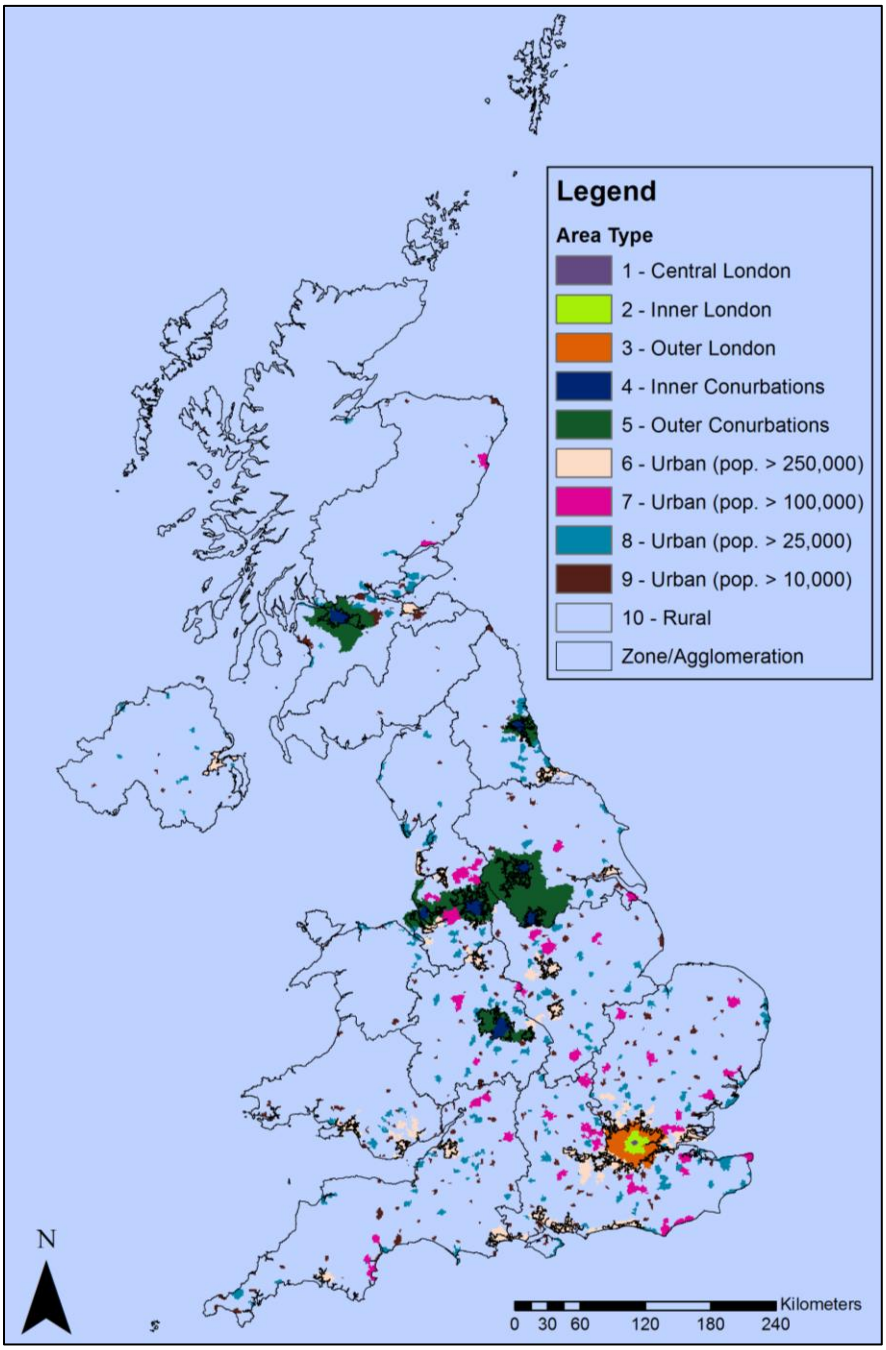
The overall damage costs of air pollutants are dominated by the contribution from long term exposure to $PM_{2.5}$ and NO_2. Damage costs per tonne of primary $PM_{2.5}$ emitted via concentration of $PM_{2.5}$ have therefore been calculated for a range of specific emission sectors and geographical locations as detailed in Table 2-2. Sector specific damage costs per tonne of NO_x emitted via concentration of NO_2 have also been calculated. Sector specific damage costs have not been calculated for the contributions of emissions to secondary $PM_{2.5}$ or ozone because the release characteristics and location of emissions are less important for these pollutants. Sector specific damage costs have not been calculated for SO_2 because the direct SO_2 impact pathways typically only make a small contribution to the overall damage costs from emission releases, which are dominated by the contribution of SO_2 emissions to $PM_{2.5}$ pathways via the formation of secondary $PM_{2.5}$.

The road transport sources are area sources and have been separated by geographical location according to ‘area types’ defined by DfT (see (Pugsley, et al., 2025). The concentrations for each sector also include the contribution from this sector to the regional primary PM concentration in addition to the local area sources. Rail and domestic combustion emissions sources have also been separated by area type. Table 2-1 lists the area types and the locations are shown in Figure 2-1. A detailed list of the assignment of urban areas to area type is also available (DfT, 2023).

Table 2-1 DfT Area types

DfT Area type	Region
1	Central London
2	Inner London
3	Outer London
4	Inner Conurbations
5	Outer Conurbations
6	Urban (population > 250,000)
7	Urban (population > 100,000)
8	Urban (population > 25,000)
9	Urban (population > 10,000)
10	Rural

Figure 2-1 Map of UK area types



The sector specific relationship between concentrations for NO_2 and emissions for NO_x have been calculated by multiplying the ugm^{-3} of NO_x per tonne of NO_x emitted for each sector by the total UK population-weighted mean of NO_2 from all sources divided by the by the total UK population-weighted mean of NO_x from all sources.

Table 2-2 Sectors for primary $PM_{2.5}$ via $PM_{2.5}$ concentrations and NO_x via NO_2 concentrations

Sector
All Sectors (National)
Industry (area sources)
Commercial
Domestic
Solvents
Road Transport Average
Aircraft
Off-road mobile machinery
Rail Average
Ships
Waste
Agriculture
Other
Road Transport Central London
Road Transport Inner London
Road Transport Outer London
Road Transport Inner Conurbation
Road Transport Outer Conurbation
Road Transport Urban Big
Road Transport Urban Large
Road Transport Urban Medium
Road Transport Urban Small
Road Transport Rural
Rail Central London
Rail Inner London
Rail Outer London
Rail Inner Conurbation
Rail Outer Conurbation
Rail Urban Big
Rail Urban Large
Rail Urban Medium
Rail Urban Small
Rail Rural
Domestic Central London
Domestic Inner London
Domestic Outer London
Domestic Inner Conurbation
Domestic Outer Conurbation
Domestic Urban Big
Domestic Urban Large
Domestic Urban Medium
Domestic Urban Small
Domestic Rural

2.2.3 Damage costs for Part A processes

The release characteristics and location of releases in relation to centres of population are particularly variable for large industrial processes. These large industrial processes are known as Part A processes and the emissions are regulated by national regulators (The Environment Agency in England, Natural Resources Wales, The Scottish Environment Protection Agency and Department of Agriculture, Environment and Rural Affairs in Northern Ireland). We have therefore calculated damage costs for nine categories of Part A processes in order to account for differences in chimneystack heights and population density. The categories are summarised in Table 2-3.

Table 2-3 Part A categories for primary $PM_{2.5}$ via $PM_{2.5}$ concentrations and NO_x via NO_2 concentrations

Average population density (persons per km^2)*	Stack Height <= 50 m and all small points	Stack Height > 50, <= 100 m	Stack Height > 100 m
<= 250	**Part A category 1**	**Part A category 4**	**Part A category 7**
> 250, <= 1000	**Part A category 2**	**Part A category 5**	**Part A category 8**
> 1000	**Part A category 3**	**Part A category 6**	**Part A category 9**

These damage costs have been derived in the same way as the rest of the sector specific damage costs (by dividing the total contribution to UK population-weighted concentrations from modelled sources within each category by the sum of emissions from the sources in each category). Note that the population density has been calculated for different areas for each stack height range. The areas are listed in Table 2-4.

Table 2-4 Population density areas for Part A categories

Stack Height <= 50 m and all small points	Stack Height > 50, <= 100 m	Stack Height > 100 m
11 km x 11 km	21 km x 21 km	31 km x 31 km

2.3 PCM model emission sensitivity coefficients method for contribution to secondary $PM_{2.5}$

The PCM model has been used to calculate the impact of NO_x emissions on ambient NO_2 concentrations and of SO_2 emissions on ambient SO_2 concentrations. These µgm^{-3} per tonne have been used in the impact pathways for NO_2 and SO_2 concentrations. Emissions of NO_x, SO_2 and NH_3 also contribute to damage costs via the secondary inorganic aerosol (SIA) contribution to ambient PM concentrations and the long- and short-term exposure to PM concentration pathways. The PCM model emission sensitivity coefficients method has been used to calculate µgm^{-3} SIA changes per tonne of NO_x, SO_2 or NH_3 emitted.

SIA within the PCM model consists of SO_4, NO_3 and NH_4 and some additional counter ions and bound water. For compliance assessment modelling the concentrations of these components are derived within the model from ambient measurement data for SO_4, NO_3 and NH_4 by interpolation and application of appropriate scaling factors, as described by (Pugsley, et al., 2025).

Results from the EMEP model have been used to calculate emission sensitivity coefficients for the UK on a 50 x 50 km grid. The coefficients represent the proportional change in UK concentrations for the SIA species for changes in UK NO_x, SO_2 and NH_3 emissions. Coefficients have also been determined for the impact of changes in emissions in the rest of the EU, emissions from other countries and emissions from shipping but these are not required for the damage cost calculations. Emission sensitivity coefficients are required because the relationship between precursor emissions and SIA concentrations is complex and the change in concentrations is typically smaller than a 1 to 1 reduction in line with changes in emissions. There are also some complex effects such as changes in NO_x emissions potentially leading to small changes in SO_4 concentrations as a result of the complex atmospheric chemistry. The emission sensitivity coefficients provide a method of capturing these complexities in the results from chemistry transport models (the EMEP model in this instance) and parameterising them in such a way that they can be used in these damage cost calculations and other applications of the PCM model, such as projections for future years.

The emission sensitivity coefficients have been used to calculate the impact of 10% reductions of UK NO_x, SO_2 and NH_3 emissions in turn on population-weighted mean annual mean SIA concentrations in the UK. 10% reductions were chosen since changes in emissions of this magnitude should result in approximately linear responses within the EMEP model, which means that the emission sensitivity coefficients should be valid for this scale of reduction. The µgm^{-3} SIA (and thus PM) per tonne change in emissions was then calculated by dividing these changes in SIA concentrations by 10% of the UK total emission for these gases that are relevant to the formation of SIA.

2.4 Specific changes to the $PM_{2.5}$, NO_2 and SO_2 concentration modelling for this update

There are few systematic changes to the PCM dispersion modelling for 2023 compared to the dispersion modelling for 2020 (which was used for the 2023 damage cost update). The following changes may be relevant to the damage cost emissions to concentrations modelling:

- Scaling factors were applied in the 2020 modelling to adjust non-road transport emissions from 2019 values to values suitable for use in dispersion modelling for 2020 to take account of the reduction in activity due to the covid-19 pandemic. This is unlikely to have a large impact on the emission to concentrations relationships and equivalent scaling is not required for the 2023 reference year modelling. Estimates of emissions for 2023 were calculated from 2022 values in the 2022 NAEI using emissions projections.

- The methods used to map the emissions estimates across the UK are subject to routine updates and improvements. These updates include changes to the distribution grids and emission factors for domestic combustion of solid fuels, the use of gas use data at a postcode level for non-domestic combustion and revised distribution grids for rail and for road transport cold start emissions. The changes in methods between the 2019 and 2022 inventories that are not expected to have large systematic impacts on the emission to concentrations relationships.

- The dispersion modelling for 2023 used estimates of domestic emissions that were calculated from estimates for 2022 from the 2022 NAEI by the application of degreeday values to account for the difference in temperatures between the two years. The degreeday values were calculated using meteorological data from the WRF meteorological model for 50 km squares across the UK. Previous assessments used a single representative degreeday scaling factor for the UK. This change is not expected to have a large impact on the emission to concentrations relationships.

2.5 PM conversion factors

Note that for these damage costs the change in $PM_{2.5}$ emission is the preferred metric for PM emissions. This is consistent with the approach adopted for the 2023 damage costs. The CRF for mortality associated with long-term exposure is for the impact of changes in $PM_{2.5}$ concentrations. Likewise, all pathways extracted from the PHE model associated with particulate matter are also expressed as $PM_{2.5}$.

The CRFs for chronic bronchitis associated with PM are for PM_{10}, rather than for $PM_{2.5}$. For ease of use and given the dominant contribution of the mortality associated with long-term exposure pathway to the total damage costs, it is recommended that all changes in PM emissions valued using these updated damage costs are expressed as changes in $PM_{2.5}$ emissions.

Damage costs have been calculated for changes in $PM_{2.5}$, therefore an adjustment is made to the PM_{10} pathways included within the damage costs calculations, which scales the primary $PM_{2.5}$ emissions to calculate an estimate of PM_{10} emissions such that the change in emissions is expressed correctly for these pathways. Ratios have been calculated from the emissions for 2023 calculated from the 2022 NAEI for each relevant sector and area type – these are presented in the table below. The value for UK total emissions ($PM_{2.5}/PM_{10}$) is 0.506. Sector specific ratios have been used for the individual emissions sectors and these vary from 0.177 for agriculture to 0.977 for off-road mobile machinery. These calculations are part of the process to calculate the damage costs. The user of the damage costs should estimate changes in emissions of $PM_{2.5}$.

Table 2-5 $PM_{2.5}$-to-PM_{10} adjustment ratios

PM Damage cost area type	$PM_{2.5}/PM_{10}$ adjustment ratio	PM Damage cost area type	$PM_{2.5}/PM_{10}$ adjustment ratio
PM	0.506	Road Transport Central London	0.551
Part A Category emissions		Road Transport Inner London	0.537
Part A Category 1	0.807	Road Transport Outer London	0.541
Part A Category 2	0.850	Road Transport Inner Conurbation	0.557
Part A Category 3	0.795	Road Transport Outer Conurbation	0.563
Part A Category 4	0.782	Road Transport Urban Big	0.561
Part A Category 5	0.761	Road Transport Urban Large	0.562
Part A Category 6	0.781	Road Transport Urban Medium	0.562
Part A Category 7	0.792	Road Transport Urban Small	0.563
Part A Category 8	0.757	Road Transport Rural	0.570
Part A Category 9	0.616	Rail Transport Central London	0.966
Area source sector emissions		Rail Transport Inner London	0.966
Industry (area)	0.220	Rail Transport Outer London	0.965
Commercial	0.929	Rail Transport Inner Conurbation	0.966
Domestic	0.977	Rail Transport Outer Conurbation	0.957
Solvents	0.633	Rail Transport Urban Big	0.961
Road Transport	0.564	Rail Transport Urban Large	0.906
Aircraft	0.636	Rail Transport Urban Medium	0.938
Offroad	0.909	Rail Transport Urban Small	0.868
Rail	0.888	Rail Transport Rural	0.839
Ships	0.947	Domestic Central London	0.987
Waste	0.923	Domestic Inner London	0.980
Agriculture	0.177	Domestic Outer London	0.978
Other	0.899	Domestic Inner Conurbation	0.977
		Domestic Outer Conurbation	0.977
		Domestic Urban Big	0.977
		Domestic Urban Large	0.976
		Domestic Urban Medium	0.977
		Domestic Urban Small	0.976
		Domestic Rural	0.977

2.6 CMAQ method for impact of changes in NOx and VOC emissions on O_3 concentrations

2.6.1 Model experiment design

Given the complexity of atmospheric chemistry, numerical models, such as atmospheric chemistry transport models (CTMs), are commonly used to simulate the processes involved and estimate the outcomes of plausible control strategies. The Community Multiscale Air Quality (CMAQ) model, developed and distributed by the US Environmental Protection Agency (EPA) is a state-of-the-science numerical air quality model with comprehensive representations of the emission, transport, formation, destruction, and deposition of many air pollutants, including NO_x/NO_2 and VOC[3].

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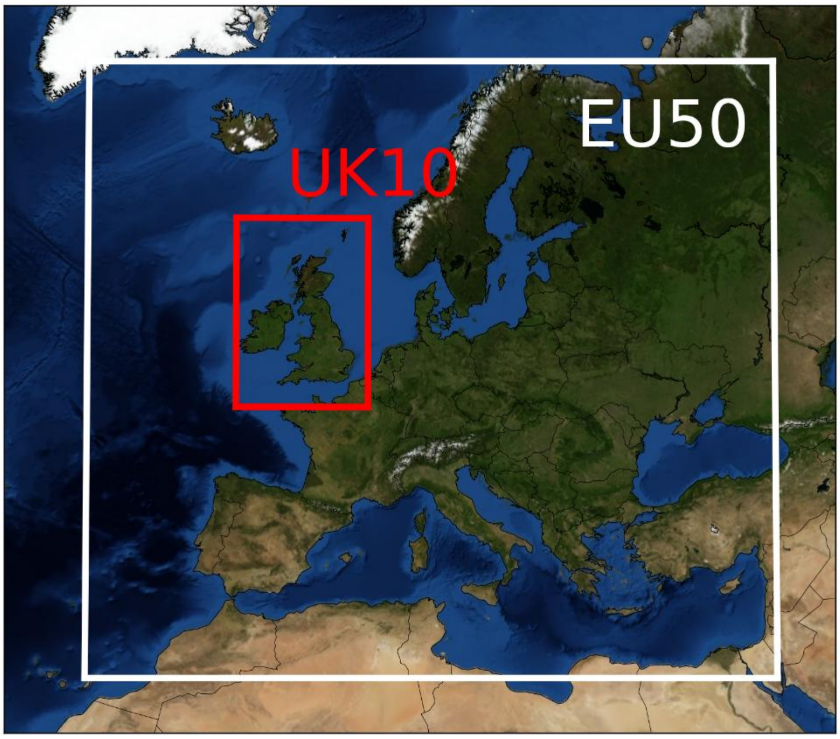
[3] Throughout this report VOC means Non-Methane Volatile Organic Compounds.

CMAQ is used to study the relationships between emissions (e.g. NO_x) and atmospheric concentrations since it simulates the main processes influencing the fate of atmospheric pollutants (turbulent dispersion, atmospheric chemistry, cloud processes, long-range transport, wet and dry deposition, etc.).

Thus, for this project we have used CMAQ version 5.4[4] with Weather Research and Forecasting (WRF) version 4.5 for the regional scale air quality and meteorological simulations, respectively. This combination has been chosen due to high level of scientific rigour. The modelling was driven by a series of 3D emission grids.

Both models follow the nested approach, i.e. dividing a large geographic domain into smaller, nested grids, with each grid representing a progressively finer level of detail. This hierarchical structure allows us to capture more detailed variations in emissions and meteorology. The modelling domain is divided into two: one covering Europe at 50km resolution and one over the UK at 10 km resolution as shown in Figure 2-2.

Figure 2-2 Regional nested modelling domains



Regional nested modelling domains shown with the matplotlib NASA ‘Blue Marble’ image used as illustration. The white box corresponds to the European domain at 50 km × 50 km horizontal resolution (EU50) and the red box to the UK domain at 10 km × 10 km horizontal resolution (UK10).

The selected meteorological year feeding the air quality simulation was 2022, corresponding to the same year as the emission inventories (Section 2.6.2.1).

The simulation starts with a spin-up period for the meteorology, followed by a 2-week period of spin-up for CMAQ (December 2021). The simulations have been defined as “forecast-cycling experiments”; i.e.,

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[4] US EPA CMAQ; https://zenodo.org/records/7218076

the calculated fields have been used to initialize the successive (following day) simulation (Morcrette, et al., 2009).

The chemical mechanism used for the gaseous species is the carbon bond mechanism (CB06r5) (Luecken, Yarwood, & Hutzell, 2019) combined with the aerosol mechanism using the 7th generation aerosol module (AERO7) (Pye, et al., 2017)

2.6.2 Emissions

2.6.2.1 Anthropogenic Emissions

The study has focused on the change on UK VOC and NOx emissions compared to a baseline. For this analysis, only the UK emissions (within the UK10 CMAQ domain) have been scaled down by 10% to analyse the reduction of these emissions by 10%.

The baseline for the EU50 domain has been based on the European Monitoring and Evaluation Programme (EMEP)[5] gridded emissions for 2022. The sectoral splits defined by EMEP have been used and temporally allocated to the emissions by sector. The speciation profiles required by CMAQ have also been used.

Anthropogenic emissions over the UK, including from agriculture, are based on the gridded emissions from the UK National Atmospheric Emission Inventory (NAEI) for 2022. The 1 km × 1 km grids were resampled into 10 × 10 km emission grids for the UK10 domain. The 2022 large point source emission inventory has also been used to vertically allocate the emissions in the CMAQ grid.

2022 was chosen as the reference year for the O_3 modelling for both the meteorological and emissions data.

2.6.2.2 Natural Sources

CMAQ also requires emissions from biogenic sources to properly account for chemical reactions within the simulations. The calculation of biogenic emissions has used an online module incorporated in CMAQ. This uses the Model of Emissions of Gases and Aerosols from Nature (MEGAN) model (version 3.2) (Guenther, et al., 2020).

CMAQ is able to calculate online windblown dust emissions based on the WRF meteorology data which are provided, and the online sea spray emissions. The sea spray emissions required the creation of ocean masks which guide the model to determine the fraction of sea water in each grid cell for both model domains.

2.6.2.3 Output metrics

The damage costs require the calculation of several ozone metrics:

- 24-hour mean O_3 concentrations averaged over the calendar year (annual mean), calculated as a population-weighted mean

- Daily maximum of 8 hour mean (annual mean of the maximum of the 24 possible 8-hour running mean concentrations in each day)), calculated as a population-weighted mean

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[5] EMEP gridded-emissions; https://www.ceip.at/the-emep-grid/gridded-emissions, 2022

- 24-hour mean O_3 concentrations averaged over a seven-month growing season from 1st March to 30th September), calculated as an area-weighted mean
- POD_6wheat (mmol m^{-2}, the annual phytotoxic ozone dose for wheat with a threshold flux of 6 nmol m^{-2} s^{-1}), calculated as an area-weighted mean

These metrics were calculated from the hourly CMAQ outputs for the base case and both of the scenarios (10% of the UK total NO_x and VOC emissions). Further details on the calculation of the metrics that are relevant to ecosystem impacts are given in section 2.7 below.

The µg m^{-3} (or mmol m^{-2}) changes per tonne changes in emissions were then calculated by dividing the changes in the population-weighted mean or area-weighted mean ozone metrics by 10% of the UK total NO_x and VOC emissions.

The relationships between NO_x emissions and VOC emissions and O_3 concentrations are complex and non-linear. However, for the purposes of calculating per tonne damage costs, both relationships have been assumed to be linear.

2.7 Calculation of O_3 metrics for estimation of ecosystem impacts

To assess the value of O_3 impact on ecosystems, specific O_3 concentration metrics were required. These were:

- POD_6wheat (mmol m^{-2}, the annual phytotoxic ozone dose for wheat with a threshold flux of 6 nmol m^{-2} s^{-1})
- 24-hour mean O_3 concentrations averaged over a seven-month growing season from 1st March to 30th September 2022.

The calculation methodology for each metric is described below.

2.7.1 Growing season average

The 7-month 24-hour mean metric was calculated directly from hourly O_3 concentrations output by the nested CMAQ model for each scenario. The impact of the two scenarios (10% reduction in UK NO_x emissions & 10% reduction in UK VOC emissions) on area-weighted 7-month 24-hour mean concentrations were then calculated from the results.

2.7.2 POD_6wheat

The POD_6wheat metric relates to the flux of O_3 through stomata rather than direct concentrations. This is dependent on a combination of the species' inherent characteristics (such as stomatal density and aperture size) and external conditions in addition to the canopy-level O_3 concentration, which is assumed to represent a reasonable estimate of the concentration at the upper surface of the laminar layer for a sunlit upper canopy flag leaf.

POD_6wheat calculations for the 2025 damage cost calculations were carried out using an updated version of Ricardo's Surface Ozone Flux Model (SOFM2). This model calculates stomatal fluxes for wheat from O_3 concentrations using the detailed methods and parameters provided in the latest revision of Chapter 3 of the LRTAP Convention Modelling and Mapping manual, published in 2017 (LRTAP, 2017), and output from the WRF and CMAQ models described in Section 2.4.

Following this methodology, stomatal flux of O_3 (F_{st}, nmol O_3 m^{-2} PLA s^{-1} where PLA is the projected leaf area) was calculated from the hourly O_3 concentration at canopy height ($c(z_1)$), the flag leaf stomatal conductance (g_{sto}, flag leaf stomatal conductance) and resistance parameters representing quasi-laminar resistance (r_b) and leaf surface resistance (r_c) using Equation 2.7.2.1.

Equation 2.7.2.1

$$F_{st} = c(z_1) \times g_{sto} \times \frac{r_c}{r_b + r_c} \ (nmol\, m^{-2}\, PLA\, s^{-1})$$

Stomatal conductance estimates the rate of gas exchange (primarily CO_2 uptake and water vapour loss but also other gases including O_3). Plants control the degree of stomatal opening in order to regulate transpiration; stomata must open to allow the gas exchange of carbon dioxide and oxygen for efficient photosynthesis, so opening is triggered by light. However, this also leads to water vapour loss, so plants maintain a balance between gas exchange and water loss. Water stress and suboptimal temperatures can therefore cause stomata to close, reducing O_3 flux.

The multiplicative algorithm of Emberson et al. (Emberson L. A.-P., 2000a) was used to calculate the stomatal conductance. This algorithm is included in the DO3SE model (Büker, 2015) ; (Emberson L. A.-P., 2001), (Emberson L. B., 2007) and is incorporated within the EMEP O_3 deposition module (Simpson, 2012) . Species-specific parameters for Atlantic bread wheat were taken from the Mapping Manual.

Equation 2.7.2.2

$$g_{sto} = g_{max} \times min(f_{phen}, f_{O3}) \times f_{light} \times max[f_{min}, (f_{temp} \times f_{VPD} \times f_{PAW})]$$

Where g_{max} is the species-specific maximum g_{sto}, (500 mmol O_3 m^{-2} PLA s^{-1} for wheat). The parameters leaf f_{phen}, fO_3, f_{light}, f_{temp}, f_{VPD}, and f_{PAW} account for the effect of phenology, accumulated O_3 exposure, light availability for photosynthesis, temperature, vapour pressure deficit (VPD), and root zone plant available water content (PAW) on stomatal conductance. f_{min} is the fractional minimal daylight g_{sto}. These functions have values ranging from 0 to 1.

SOFM2 was updated to use outputs from the WRF and CMAQ models across the model domain, including hourly mapped O_3 concentrations at canopy height, meteorological data, and soil parameters:

- Meteorological parameters including temperature, wind speed, friction velocity, and solar radiation were taken from output from the WRF simulations described in Section 2.4.
- To allow calculation of the Plant Available Water content, soil parameters were taken from the NOAH Land Surface Model used in the WRF and CMAQ simulations and volumetric soil moisture was extracted from WRF output.
- Growing season dates were calculated across the model domain using an approach based on accumulated thermal time as described in the Mapping Manual. Following this approach, the growing season occurs later in the year and is longer in colder regions (i.e. further north and in areas with significant elevation).

POD_6wheat was calculated from the stomatal O_3 flux as the integrated total flux above a threshold of 6 mmol m^{-2} PLA in daylight hours throughout the year.

Equation 2.7.2.3

$$POD_6 wheat = \sum [(F_{st} - 6) \times (\frac{3600}{10^6})] \ (mmol\, m^{-2}\, PLA)$$

This separate SOFM2 post-process was run on the base case hourly CMAQ model results on a 10 km x 10 km UK grid and the two scenarios representing a 10% reduction in UK NOx emissions and UK

VOC emissions, respectively. The impact of these scenarios on area weighted mean POD_6wheat were then calculated from the results.

3 Estimation and valuation of human health impacts

3.1 Introduction

For the 2025 damage cost update, changes were made to the input data underpinning the calculation of the human health impacts associated with exposure to air pollution. The approach for this update, relative to previous 2023 update, is summarised in Section 3.2 and explained in further detail in the remainder of Sections 3 and 4.

3.2 Description of changes in approach for this update

The approach adopted for this update to the damage costs largely follows the methods and assumptions used for the damage cost update published in 2023. The 2025 update has been based upon the current evidence base and discussions with the IGCB. Primarily, the focus of the 2025 update has been to review and draw on more up-to-date evidence to capture the latest understanding of the relationship between air pollution and health impacts.

Since the 2023 update, the World Health Organisation (WHO) has launched a new project called Estimating the Morbidity from Air Pollution and its Economic Costs (EMAPEC)[6]. A report has been published summarising Concentration Response Functions (CRFs) suitable for health impact assessment (Forastiere, 2024). This report potentially provides suitable updated CRFs for existing morbidity pathways within the damage costs. The report provides CRFs for most, but not all, of the morbidity end points currently included in the analysis underpinning the damage costs. This also includes CRFs for some health endpoints not currently included in this analysis. These CRFs have not yet been formally endorsed by the WHO. COMEAP has also not yet reviewed this work. It is likely that these new CRFs could be included in the next update of damage costs if COMEAP recommends their use. In most cases, there is not a direct one-to-one match between the current damage cost pathways CRFs and those available from Forastiere (2024).

The WHO Update of the Health Risks of Air Pollution in Europe (HRAPIE-2) project has provided updated CRFs for mortality that could be used. Orellano et al (2024) provides a CRF for $PM_{2.5}$ and Kasdagli et al (2024) provides CRFs for long term exposure to NO_2 and O_3 (Orellano et al., 2024; Kasdagli et al., 2024). However, more work by COMEAP is needed to decide how the new values could be used within a damage cost update, particularly around adjustment for overlap between $PM_{2.5}$ and NO_2. The current damage cost method reduces the NO_2 coefficient to 40% of the single pollutant CRF value in order to take overlap into account. Chen et al (2024) provides a possible method to adjust single pollutant CRFs by using results from two pollutants models (Chen et al. , 2024). More work is needed before such an approach could be used within a formal update of the damage costs.

The following updates were made to the health impact analysis (and workbook capturing this):

- The population data was updated to reflect the latest ONS population estimates (ONS, 2024).
- England, Wales and Northern Ireland baseline mortality rates and hospital admission rates were updated in line with the latest information available (ONS, 2024; NISRA, 2024; National Records of Scotland, 2024)

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[6] WHO EMAPEC; https://www.who.int/activities/estimating-the-morbidity-from-air-pollution-and-its-economic-costs

3.3 Concentration response functions (CRFs) for health outcomes

3.3.1 CRFs carried forward from previous COMEAP guidance

The health impacts of air pollution are estimated based on Concentration Response Functions (CRFs). CRFs capture the relationship between human exposure to an air pollutant and the consequent health implications, which are expressed as change in a health (or non-health) outcome for a given change in air pollutant concentrations.

In 2023, the Interdepartmental Group on Costs and Benefits (IGCB) recommended the set of CRFs that should be used for the appraisal of air quality impacts (Defra, 2013). These CRFs were taken from an extensive underlying literature review on the health effects of air pollution and follow the recommendations of COMEAP (COMEAP, 1998; COMEAP, 2009; COMEAP, 2010). The health impact pathways included in the 2013 guidance have been carried forward as part of this 2025 damage cost update.

COMEAP has subsequently published additional reports recommending health impact pathways for inclusion in the appraisal of air pollutant impacts (and the appropriate methodology for doing so). This includes:

- Impacts of ozone exposure on hospital admissions and mortality attributable to short-term exposure[7] (COMEAP, 2015b).
- Statement on quantifying mortality associated with long-term average concentrations of fine particulate matter ($PM_{2.5}$) (COMEAP, 2018b)

The CRFs used for the estimation of this 2025 damage cost update are set out in Table 3-1 below.

Table 3-1 CRFs applied in the updated damage costs (% change or Odds Ratio per 10 μg/m³ change in concentration for relevant averaging period) – COMPEAP

				% or Odds ratio change per μg/m³ change in pollutant		
Pollutant	**Pathway**	**Air pollution metric**	**CRF type**	**Low**	**Central**	**High**
SO_2	Mortality attributable to short-term exposure (1)	Annual average	Relative Risk (RR), %	0.6	0.6	0.6
SO_2	Respiratory hospital admission (1)	Annual average	RR, %	0.5	0.5	0.5
O_3	Mortality attributable to short-term exposure (2)	Daily maximum of 8 hour mean	RR, %	0.12	0.34	0.56
O_3	Respiratory hospital admission (2)	Daily maximum of 8 hour mean	RR, %	0.3	0.75	1.2
O_3	Cardiovascular hospital admission* (2)	Daily maximum of 8 hour mean	RR, %	-0.06	0.11	0.27
NO_2	Mortality associated with long-term exposure (3)	Annual average	RR, %	0.8 (0.2)	2.3 (0.92)	3.7 (2.035)
PM_{10}	Chronic bronchitis* (4)	Annual average	Odds ratio (OR)	1.02	1.32	1.71

Notes: (*) marks pathways that are only included in the high sensitivity scenario. CRFs with the adjustment for overlap with $PM_{2.5}$ applied are included in brackets. Sources: (1) (Defra, 2013), (2) (COMEAP, 2015b), (3) (COMEAP, 2018) (4) (COMEAP, 2016), (5) (PHE, 2018)

While there are CRFs for O_3, these are only relevant for the damage costs associated with NO_x and VOC emissions because O_3 is a secondary air pollutant, for which there are no emissions.

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[7] Described previously as deaths brought forward

The IGCB's guidance did not include a sensitivity range for CRFs linking short-term, acute exposure to SO_2 to hospital admissions or mortality. Hence, the CRFs used to assess these impacts are not adjusted in the derivation the 'low' and 'high' damage cost sensitivities.

In the previous estimation of damage costs (prior to 2023), impacts on health from ozone exposure were estimated using a range of thresholds, where a threshold represents a minimum level of concentration that must be reached before impacts on health start to occur. For this 2025 update, we have not applied a threshold to the calculation of effects across all damage cost sensitivities, based on the most recent advice from COMEAP regarding the estimation of effects associated with ozone exposure (COMEAP, 2015).

3.3.2 CRFs carried forward from PHE (morbidity associated with long-term exposure)

In the 2020 damage cost update, pathways for five chronic diseases (asthma in adults, asthma in children, coronary heart disease, stroke, diabetes type 2 and lung cancer) explored by Public Health England (PHE, 2018) were included for the first time. CRFs in relation to these new morbidity pathways for a NO_2 and $PM_{2.5}$ were extracted from the report published by Public Health England (PHE), underpinned by scientific evidence.

The PHE CRFs that we have used in the updated damage costs are summarised in Table 3-2 below. We have included only the pathways that are considered more certain in the central damage costs, further detailed in Table 3-3 in Section 3.3.3. The CRFs for coronary heart disease (CHD) and stroke from the PHE work were not used in the 2023 or 2025 updates to the damage costs because they have been replaced by updated CFRs for ischemic heart disease (IHD) and stroke.

Table 3-2 CRFs applied in the updated damage costs (% change or Odds Ratio per 10 µg/m^3 change in concentration for relevant averaging period) – PHE morbidity pathways

				% or Odds Ratio change per 10 µg/m^3 change in pollutant		
Pollutant	**Pathway (Source)**	**Air pollution metric**	**CRF type**	**Low**	**Central**	**High**
$PM_{2.5}$	Diabetes Type 2 (Eze, et al., 2015)	Annual average	Relative Risk (RR), %	2.00	10.00	18.00
NO_2	Diabetes Type 2 (Eze, et al., 2015)	Annual average	RR, %	2.00	5.00	7.00
$PM_{2.5}$	Lung cancer (Hamra, et al., 2015)	Annual average	RR, %	4.00	9.00	14.00
NO_2	Lung cancer (Hamra, et al., 2015)	Annual average	RR, %	0.00	2.00	3.00
NO_2	Asthma (Adults) (Jacquemin, et al., 2015)	Annual average	Odds Ratio (OR)	1.00	1.04	1.08
$PM_{2.5}$	Asthma (Older Children) (Khreis, et al., 2016)	Annual average	OR	1.22	1.48	1.97
NO_2	Asthma (Small Children) (Khreis, et al., 2016)	Annual average	OR	1.01	1.08	1.12
NO_2	Asthma (Older Children) (Khreis, et al., 2016)	Annual average	OR	1.00	1.03	1.06

All CRFs are assumed to represent a change in incidence, as suggested by most of the references that were used in the PHE report. In addition, the NO_2 CRFs were adjusted by applying a factor of 40% to take account of overlaps between risks produced by $PM_{2.5}$, in order to be consistent with the approach adopted by PHE.

PHE's model also included impacts on low birth weight and dementia. However, following discussion between Defra and PHE there was some concern regarding the inclusion of these pathways, and they were deprioritised relative to the inclusion of the other pathways, and not included in this round of updates.

3.3.2.1 Baseline epidemiological data

Baseline epidemiological data for the diseases of interest were extracted from (PHE, 2018), which is the latest tool published by Public Health England. The data remains consistent with the 2023 update This compiles information from numerous sources:

- *Asthma in adults.* Incidence data for age groups older than 16, both genders. British Lung Foundation (BLF) statistics sourced from The Health Improvement Network (THIN) database[8].
- *Asthma in children.* Incidence data for small (≤ 6 years old) and older children (7-15 years old). British Lung Foundation (BLF) statistics sourced from The Health Improvement Network (THIN) database.
- *Coronary heart disease (CHD).* Incidence data for all age groups, male and female. Data sourced from the British Heart Foundation (BHF) cardiovascular disease statistics 2014.
- *Stroke.* Incidence data for all age groups, male and female. Data sourced from the British Heart Foundation (BHF) cardiovascular disease statistics 2014.
- *Diabetes Type 2.* Incidence data for age groups older than 20, male and female. Data sourced from the National Diabetes Audit 2015-2016.
- *Lung cancer.* Incidence data for all age groups, male and female. Data sourced from Cancer Research UK (2012-2014).

More detailed information on the sources of baseline prevalence and incidence rates for morbidity effects associated with long-term exposure is provided in Section 3.5.

The data provided above was per 100,000 persons of each age group. To obtain an age- and gender-weighted incidence of a disease *i*, Equation 3.3.2.1 was applied:

Equation 3.3.2.1

$$I_i = \sum_j^J \sum_k^K \frac{N_{j,k}}{N} \cdot I_{i,j,k}$$

Where:

I_i is the age- and gender-weighted incidence of a disease *i*.

$N_{j,k}$ is the population of age group *j* and gender *k* in the United Kingdom.

N is the total population of the United Kingdom.

$I_{i,j,k}$ is the incidence of disease *i*, age group *j* and gender *k*.

3.3.2.2 Calculation of the change in incidence

The estimation of the change in incidence due to a decrease of 1 µg/m^3 of $PM_{2.5}$ or NO_2 is different depending on whether the CRF is based on the Relative Risk, Hazard Ratio or Odds Ratio.

Relative risks and Hazard ratios

The change in incidence (ΔI_i) per 100,000 inhabitants when the CRF is based on either the Relative Risk or Hazard Ratio is estimated as the product between the concentration of the pollutant, the baseline incidence, and the population as in Equation 3.3.2.2:

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[8] Linearity on the log scale: log-linearity. Cohort studies of mortality typically relate the natural log of the hazard function to the concentration. In practice, for a small hazard ratio (as found in most air pollution studies) and over a small concentration range (as is usually the case in a health impact assessment) there is little difference between a linear and log-linear relationship. This might not be the case when larger concentration differences are being considered

Equation 3.3.2.2

$$\Delta I_i = \frac{\Delta C_{Pol}}{C_{Inc}} \cdot RR \cdot \frac{N}{10^5} \cdot I_i$$

Where:

ΔC_{Pol} is the concentration of a given pollutant ($PM_{2.5}$, NO_2).

C_{Inc} is the concentration increment on which the CRF is based (5 or 10 μg/m^3).

RR is the Relative Risk (or Hazard ratio, if applicable), expressed as a percentage increase in the effect per change in increment.

N is the total population of the United Kingdom.

I_i is the age- and gender-weighted incidence of a disease *I,* per 100,000 people.

Odds Ratio

The estimation of the change in incidence (ΔI_i) per 100,000 inhabitants when the CRF is based on the Odds Ratio (OR) is more complex, as it requires an estimate the odds of reporting the disease at the new concentration (κ_i) first, as in Equation 3.3.2.3:

Equation 3.3.2.3

$$\kappa_i = \exp\left(-\ln(OR) \cdot \frac{\Delta C_{Pol}}{C_{Inc}} + \ln\frac{I_i}{10^5 - I_i}\right)$$

The change in incidence (ΔI_i) per 100,000 inhabitants can be then estimated as a function of the odds of reporting the disease at the new concentration (κ_i) as in Equation 3.3.2.4:

Equation 3.3.2.4

$$\Delta I_i = \frac{N(1+\kappa_i)}{\kappa_i(I_i - 1) + I_i}$$

In the case where relative risk values were based on concentration increments of 5 μg/m^3 (C_{inc}), these were used in preference to those extrapolated in the PHE report to a 10 μg/m^3 concentration increment base. This was done in order to be consistent with the methodology explained above, since the extrapolation of relative risk values made in the PHE report was non-linear and the damage cost approach assumes a linear scaling.

3.3.3 Health impact pathway summary

The health impact pathways and Concentration Response Functions captured in the 2025 updated damage costs remain the same as those used in the previous damage costs, updated with COMEAP publications. The CRFs used for the estimation of the updated damage costs for the health pathways (apart from for the productivity pathways, which are summarised in (Ricardo-AEA, 2014)– see section 3.6) are set out in Table 3-3 below.

Please note that annual average exposures are used to assess the effects that are associated with short-term average concentrations. The sum of short-term effects calculated on a daily basis from a combination of daily means over the period of a year and the CRF for daily effects is mathematically the same as a calculation based on an annual mean and a duration of a year. Hence, it is more efficient and simpler to use annual mean for these calculations, whilst making no difference to the outturn results.

Please also note that the low, central and high in the table do not in all instances relate to the CRF values applied in the low, central and high damage cost sensitivities. This simply presents the confidence interval bound around each CRF presented in the underlying literature. The CRF value from the confidence interval does vary between the low, central and high damage costs; however, the inclusion of the impact pathways themselves also varies between the sensitivities. Table 6-1 presents the impact pathways that are included in each sensitivity scenario, and the CRFs that are selected from the underlying confidence interval in each damage cost value.

Finally, emissions of NO_x, SO_2 and NH_3 also contribute to damage costs via the secondary inorganic aerosol (SIA) contribution to ambient PM concentrations and the long and short-term exposure to PM concentration pathways. A full mapping of the different impact pathways included in each of the damage costs is presented in Table 3-4 also below. Primary effects, such as the mortality associated with long-term exposure to PM2.5 resulting from emissions of $PM_{2.5}$ are labelled ‘P’. Secondary effects, such as the mortality associated with long-term exposure to $PM_{2.5}$ resulting from emissions of NO_x are labelled ‘2’.

Table 3-3 CRFs applied in the updated damage costs (% change or Odds Ratio per 10 μg/m^3 change in concentration for relevant averaging period)

Pollutant	Pathway	Air pollution metric	CRF type	% or Odds ratio change per 10ugm^{-3} change in pollutant		
				Low	Central	High
$PM_{2.5}$	Mortality (long-term exposure)	Annual average	Relative Risk (RR), %	6	8	9
$PM_{2.5}$	Respiratory hospital admission	Annual average	RR	-0.63	0.96	2.58
$PM_{2.5}$	Cardiovascular hospital admission	Annual average	RR	0.26	0.9	1.53
$PM_{2.5}$	IHD incidence	Annual average	Hazard Ratio (HR)	-1.00	7.00	16.00
$PM_{2.5}$	Stroke incidence	Annual average	HR	-1.00	11.00	25.00
$PM_{2.5}$	Diabetes incidence	Annual average	RR, %	2.00	10.00	18.00
$PM_{2.5}$	Lung cancer incidence	Annual average	RR, %	4.00	9.00	14.00
$PM_{2.5}$	Asthma incidence (Older Children)	Annual average	Odds Ratio (OR)	1.22	1.48	1.97
PM_{10}	Chronic Bronchitis incidence	Annual average	RR, %	1.02	1.32	1.71
SO_2	Mortality attributable to short-term exposure	Annual average	RR, %	0.6	0.6	0.6
SO_2	Respiratory hospital admission	Annual average	RR, %	0.5	0.5	0.5
O_3	Mortality attributable to short-term exposure	Daily maximum of 8 hour mean	RR, %	0.12	0.34	0.56
O_3	Respiratory hospital admission	Daily maximum of 8 hour mean	RR, %	0.3	0.75	1.2
O_3	Cardiovascular hospital admission	Daily maximum of 8 hour mean	RR, %	-0.06	0.11	0.27
NO_2	Mortality (long-term exposure)	Annual average	RR, %	0.8	2.3	3.7
NO_2	Respiratory hospital admission	Annual average	RR, %	0.33	0.57	0.82
NO_2	Cardiovascular hospital admission	Annual average	RR, %	0.32	0.66	1.01
NO_2	Diabetes incidence	Annual average	RR, %	2.00	5.00	7.00
NO_2	Lung cancer incidence	Annual average	RR, %	0.00	2.00	3.00
NO_2	Asthma incidence (Adults)	Annual average	OR	1.00	1.04	1.08
NO_2	Asthma incidence (Small Children)	Annual average	OR	1.01	1.08	1.12
NO_2	Asthma incidence (Older Children)	Annual average	OR	1.00	1.03	1.06

Table 3-4 Mapping of primary and secondary effects against each damage cost

Pollutant		$PM_{2.5}$	PM_{10}	PM_{10}	$PM_{2.5}$	PM_{10}	SO_2	SO_2	NO_2	NO_2	NO_2	O_3	O_3	O_3	O_3	O_3	PM_{10}	SO_2	SO_2	O_3	NO_2	NH_3	$PM_{2.5}$	NO_2	$PM_{2.5}$	$PM_{2.5}$	NO_2	$PM_{2.5}$	NO_2	$PM_{2.5}$	NO_2	NO_2
Pathway		Mortality (long-term exposure)	Respiratory hospital admission	Cardiovascular hospital admission	Productivity	Chronic Bronchitis Incidence	Mortality attributable to short-term exposure	Respiratory hospital admission	Respiratory hospital admission	Mortality (long-term exposure)	Productivity	Mortality attributable to short-term exposure	Respiratory hospital admission	Cardiovascular hospital admission	Productivity	Material damage	Building soiling	Material damage	Ecosystems	Ecosystems	Ecosystems	Ecosystems	IHD Incidence	Asthma (Adults) Incidence	Stroke Incidence	Diabetes Incidence	Diabetes Incidence	Lung Cancer Incidence	Lung Cancer Incidence	Asthma (Children) Incidence	Asthma (Small Children) Incidence	Asthma (Older Children) Incidence
Damage cost	NO_x	2	2		2	2*			P	P	P*	2	2		2*	2				2	P		2	P*	2	2*	P*	2	P*	2	P	P
	SO_2	2	2		2	2*	P	P										P					2		2	2*		2		2		
	NH_3	2	2		2	2*																P	2		2	2*		2		2		
	VOC											2	2		2*	2				2												
	$PM_{2.5}$	P	P		P	P*											P						P		P	P*		P		P		

P = primary effect; 2 = secondary effect; * = impact pathways that are only included in the high sensitivity scenario.

3.4 Mortality associated with long-term exposure and life-table calculations

3.4.1 Methodology for calculating long-term air pollution impacts

For the 2025 damage cost update, the methodology for calculating long-term air pollution impacts is unchanged from the methods applied in the 2023 damage cost update.

The methodology used to calculate the impacts of long-term (or 'chronic') exposure to air pollution on mortality is known as the 'life-tables technique' and is based on a report by (IOM, 2000) and a subsequent publication by (Miller & Hurley, 2003)). In the updated damage costs, life tables are applied to calculate the mortality effects associated with long-term exposure to $PM_{2.5}$ and NO_2.

A life table is used to summarise patterns of survival in populations. Standard life-table calculations compute survival rates at different ages. It uses age-specific death rates, derived from numbers of deaths in each age group and mid-year population sizes for each age group. From these survival rates average life expectancy, from either birth or a specific achieved age, can be derived. Combining these values with numbers in the population affected allows prediction of the total numbers of life years lived at each age.

To derive health impacts associated with a change in pollutant concentrations, the basic approach for a given population is to:

- obtain information on current mortality rates
- predict future mortality using current mortality rates and assumptions about future demography using life-table calculations, in the absence of changes in air pollution
- create an alternative scenario by adjusting mortality rates according to evidence regarding the effect of pollution on mortality, leaving other baseline assumptions unchanged
- compare predicted life expectancy between the scenario without pollution changes and the alternative scenario to give estimates of the effect on the target population of the pollution change (in life-years).

Life-table calculations were undertaken by Brian Miller (Institute of Occupational Medicine, IOM) using the IOMLIFET[9] system. Calculations were based on mid-year population estimates and mortality rates for 2012[10]. These formed a baseline scenario in which it was assumed that mortality rates identified in 2012 remain constant over the assessment period and the impact of net migration does not alter population sizes or mortality rates.

Life-table calculations were undertaken for a one-year pulse reduction of $1\mu gm^{-3}$ in annual average $PM_{2.5}$ concentrations. When the damage costs were initially developed, mortality impacts associated with long-term exposure were calculated for an annual (1 year) and sustained (for 5, 20 and 100 year) pollution pulses. This update to the damage costs has only used a one-year pulse approach to be consistent with methodology underpinning the original damage costs. These costs used an annual pulse to provide flexibility in the damage cost approach: not all policies would be expected to last one year but a one-year reduction in emissions can readily be scaled up to provide an approximation for a variety of durations. As such, by using an annual pulse approach, this implicitly assumes that impacts of emissions changes are additive across different years of analysis (for example, where a policy has impacts on emissions for consecutive years, these can be added together) and in the short term the difference between assessing the impacts of a sustained change in concentrations and the sum of annual pulse changes over the same time period are negligible.

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[9] For further detail, see: http://www.iom-world.org

[10] Data for population and mortality for single year age groups up to age 90 (aggregate population and mortality rates for ages 90+ were applied to all ages over 90) were sourced from the different UK statistics authorities (ONS, GRO Scotland and NISRA) for 2012.

Calculations were undertaken for scenarios with different CRFs to reflect the low, central and high uncertainty ranges recommended by COMEAP (COMEAP, 2022). The life-table outputs for the 'alternative' scenarios (i.e. including the impact of the marginal air pollutant change on mortality rates) were compared with those for the baseline. This provided an estimate of the total life years gained for the population aged 30+ in the UK over a 100-year assessment period (an alternative scenario involving a one-year reduction is not predicted to have any impact on new birth cohorts). These results were subsequently scaled according to the ratio of CRFs to derive life-table calculations for the mortality impacts of NO_2.

In the 2025 damage cost update, an uplift has been applied to lifetable outputs to adjust for changes in population and mortality rate since the original analysis was performed. Death rates covering all ages have been used to scale the results from the life tables. This assumes that the trends in rates for ages 30+ and all ages are similar. In all instances death rates exclude deaths for external causes such as suicide, homicide and accidents.

3.4.2 Cessation lag

The potential lag between a reduction in pollutant concentrations and a change in the risk rate of a chronic health outcome is unknown. When the damage costs were initially developed, a lag range between 0 and 40 years was assumed for all mortality effects associated with long-term exposure based on the then prevailing advice of COMEAP (Department of Health, 2001). It was noted that neither a lag time of 0 nor 40 years would be likely for all affected persons, but evidence suggested that could be feasible for a proportion of deaths depending on health condition. In summary, it was assumed that the average lag time for all-cause mortality was somewhere between the two extremes. The original damage costs varied the length of lag from 40 to 0 years between 'low' and 'high' damage cost sensitivities respectively.

Cessation lag is a term used to denote the time pattern of reductions in mortality hazards following a reduction in pollution. In their 2010 report, COMEAP note that there is little direct evidence regarding cessation lags but adopted the approach agreed by the US Environmental Protection Agency (EPA) in 2004 and re-affirmed in the EPA's analysis in 2010. This approach uses a distribution of impacts on mortality rates across different lag times. Specifically: 30% of the risk reduction occurs in the first year after pollution reduction, 50% occurs across years 2-5 with the remaining 20% distributed across years 6-20 with smoothed annual values. COMEAP (COMEAP, 2022) recommended continuing to use this cessation lad approach.

COMEAP (COMEAP, 2009) considered that while, in principle, it might take 40 years for all benefits to be achieved, in practice benefits are likely to occur earlier, with a significant proportion in the first five years. As such, the three components of the cessation lag approach were considered to represent the short-term, cardiovascular and lung cancer mortality effects respectively. The most recent version of IOMLIFET permits calculations with arbitrary lag patterns and was used here to implement the EPA pattern of lags.

3.4.3 Results and interpretation

Each alternative scenario, assuming a unit reduction in pollutant concentration combined with different CRF sensitivities, is compared to the baseline scenario to derive the impact on life years. A summary of the impacts across the scenarios is presented in Table 3-5 below. The results show cumulative life years gained across all age cohorts, sexes and calendar years from 2012 to 2112. A value for a 1µgm^{-3} one-year pulse reduction in NO_2 has been calculated from this value by linear scaling, using the ratio of CRFs for mortality associated with long-term exposure to $PM_{2.5}$ and NO_2.

Table 3-5 Life years gained by UK 2023 population aged 30+ from a 1µgm^{-3} one-year pulse reduction in $PM_{2.5}$

Parameter	Result
Total life years gained (Range from low to high CRF bounds)	54,160 (41,056 – 60,323)

3.5 Baseline population and health response rates

This 2025 damage cost update has adjusted the population and baseline health outcomes data used in the calculations in line with the latest publications.

Population data for the UK and each of the Devolved Administrations was taken from ONS's mid-year population estimates for 2023 (ONS, 2024). This represents resident population, which is also used when modelling exposure. Hence the calculations were based on a UK population of around 68.3m, of which 12.9m were aged 65 and over.

Data for the number of deaths in the UK for 2023 were aggregated from data for individual Devolved Administrations sourced from the (ONS, 2024), (NISRA, 2024) and (National Records of Scotland, 2024). These data were then combined with the population data to derive a baseline mortality risk rate against which the impacts of air pollution are assessed.

Information on the number of hospital admissions per annum split by cause was also aggregated from data for each Devolved Administration: from (NHS Digital, 2024) for England, (DHSSPSNI, 2024) for Northern Ireland and NHS Wales (NHS Wales Informatics Service, 2024). No consistent data were available for Scotland hence an average risk rate was calculated based on the numbers of hospital admissions in England, Northern Ireland and Wales and it is assumed that this is a reasonable approximation for the rate across the whole of the UK. The latest data available were for the year 2023/24.

Baseline rates for the prevalence and incidence of morbidity associated with long-term exposure to air pollution were gathered from a range of sources:

- *Chronic bronchitis.* The longstanding conditions of the respiratory system and doctor-diagnosed COPD were taken from the same sources used by (COMEAP, 2016)) in their calculation of chronic bronchitis effects[11].
- *Asthma in adults.* Data for the number of people a year that receive an asthma diagnosis each year (incidence) was taken from the asthma statistics page of the British Lung Foundation website (British Lung Foundation, 2018). This data was then used to update baseline incidence of asthma in the UK.
- *Asthma in children.* Incidence data for small (≤ 6 years old) and older children (7-15 years old). British Lung Foundation (BLF) statistics sourced from The Health Improvement Network (THIN) database (British Lung Foundation, 2018).
- *Ischemic heart disease (IHD).* For the 2020 damage costs, incidence data for all age groups, male and female, for coronary heart disease was sourced from the British Heart Foundation (BHF) cardiovascular disease statistics 2014.
- *Stroke.* Incidence data for all age groups, male and female. Data sourced from the British Heart Foundation (BHF) cardiovascular disease statistics 2014.
- *Diabetes Type 2.* Incidence data for age groups older than 20, male and female. Data sourced from the National Diabetes Audit 2015-2016.
- *Lung cancer.* Incidence data for all age groups, male and female. Data sourced from Cancer Research UK (2012-2014).

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[11] COMEAP adopted baseline prevalence rates for chronic phlegm in England from the Health Survey for England and in Scotland from the Scottish Health Survey of 2011. Both sources have been updated with the latest data. For the Health Survey for England, the prevalence of longstanding conditions of the respiratory system was used in place of phlegm (HSE, 2024). In the Scottish Health Survey, doctor-diagnosed COPD was used instead of phlegm (Scottish Government, 2024). Other chronic bronchitis data from the same COMEAP (2016) report, such as the coefficients for the odds ratio of chronic phlegm in never-smokers, remain unchanged.

These data are typically presented in units per 100,000 persons of each age group. Where applicable, to obtain an age- and gender-weighted incidence of a disease *i*, Equation 3.3.2.1 was applied.

The risk rates used in the estimation of the damage costs are presented in Table 3-6 below.

Table 3-6 Health outcome risk rates used for damage cost estimation (number of cases (prevalent or incident) per 100,000 of population per annum)

Metric	Type	Risk rate all ages	Risk rate in ages 16+	Risk rate in ages 65+	Risk rate 0-5	Risk rate 6-15
Attributable Deaths	Incidence	920	-	-	-	-
Cardiovascular hospital admission	Incidence	1,054	-	3,845	-	-
Respiratory hospital admission	Incidence	1,814	-	4,744	-	-
Chronic bronchitis	Prevalence	-	6,670	-	-	-
Asthma in adults	Incidence	-	186	-	-	-
Asthma in children	Incidence	-	-	-	929	456
IHD	Incidence	176	-	-	-	-
Stroke	Incidence	137	-	-	-	-
Diabetes (Type II)	Incidence	577	-	-	-	-
Lung cancer	Incidence	80.2	-	-	-	-

3.6 Impacts on productivity

For the 2025 damage cost update, the methodology for estimating impacts on productivity is unchanged from the methods applied in the 2023 damage cost update.

(Ricardo-AEA, 2014) explored the impacts of air pollution on productivity. This paper, in particular the identification and application of relevant concentration response functions, was informed by preceding studies (Ostro, 1987).

The study developed a method to quantify these effects through five pathways. These focussed on the direct impacts of air pollution on human health via inhalation (and hence on labour as an input into production):

- Mortality (due to long-term and short-term exposure) in workforce
- Morbidity in the workforce (absenteeism)
- Morbidity in the workforce (presenteeism)
- Absence in the workforce due to morbidity in dependents
- Health impacts (mortality and morbidity) in non-market productive activities (e.g. volunteering and non-paid caring).

Eight other pathways were identified but not taken forward for quantification. These pathways included for example: impacts on visibility, animal health, and indirect impacts on human health via consumption of food or water.

The methodology to quantify the impacts under each pathway taken forward follows the widely recognised Impact Pathway Approach. The valuation of these health impacts uses the Human Capital Approach (HCA) to assess lost productivity. Under the HCA, productivity loss is measured as the length of potential productive time that the person is unable to work multiplied by a value of marginal productivity revealed in the market.

The study estimated that the burden associated with 2012 levels of pollutants had a total cost of £2.7bn (price year 2012) through its impact on productivity in that year. Some of the pathways captured in this analysis overlap with those pathways and impacts already captured in the existing damage costs and IGCB appraisal guidance. The study identified only £1.1bn (price year 2012) of these costs are additional to those that would have been captured using the existing IGCB appraisal guidance.

The updated damage costs include an estimate of the impact of air pollution on productivity following the approach described in the report. Only those impact pathways that are deemed additional to those already included in the existing damage cost estimates are added. This seeks to avoid double counting of effects (further discussion on the interaction and overlaps between these effects and those already captured by the IGCB guidance can be found in the underlying report (Ricardo-AEA, 2014)). The impact pathways included under the damage cost estimates, across the low, central and high sensitivities, are:

- absenteeism and working days lost (WDL) for employees, volunteers and carers ($PM_{2.5}$)
- presenteeism and minor restricted activity days (mRADs) for employees ($PM_{2.5}$ and O_3).

In addition, the high damage cost estimates also include impacts on school days lost (SDL) (and consequent effect of absent workers to care for dependents) through exposure to PM_{10} and O_3.

Low, central and high estimates of the additional productivity impacts described previously are included in the low, central and high damage costs respectively. Several parameters are varied to produce these different sensitivity estimates, alongside the impact pathways included as set out above. The parameters flexed under each sensitivity are set out in Table 3-7 below.

Table 3-7 Parameters flexed to produce low, central and high productivity cost estimates

Productivity impact sensitivity	Low	Central	High
Impact pathways	WDL ($PM_{2.5}$), mRADS $PM_{2.5}$ and O_3)	WDL ($PM_{2.5}$), mRADS $PM_{2.5}$ and O_3)	WDL ($PM_{2.5}$), mRADS $PM_{2.5}$ and O_3), SDL (PM_{10} and O_3)
CRF applied from CRF confidence interval	Low	Central	High
Unit values	Average wage per worker	CBI value of average lost productivity per worker	Average GDP per day worked
Baseline rates of absence	Uses only air pollutant related health impacts (e.g. respiratory or cardio-vascular complaints) to set baseline absence rates	Uses total absence rate to set baseline (i.e. covering all causes, not just air quality related complaints)	Uses total absence rate to set baseline (i.e. covering all causes, not just air quality related complaints)

3.7 Valuation of health outcomes

Finally, the health impacts from air pollution exposure are valued, monetarily. This is done by combining the estimated quantities of health effects (e.g., hospital admissions, life years, QALY, etc.) and the monetary value for each of these health effects. For these monetary values, the damage cost analysis uses the Value Of a Life Year (VOLY) to monetise mortality impacts; the value of a QALY to monetise morbidity impacts, and the value of hospital admissions to monetise impacts in hospital activity. The values used in the 2025 damage cost update are the same as those used in the calculation of the original damage costs, with pricing adjustments and targeted improvements.

3.7.1 Mortality associated with long-term exposure (VOLY)

The unit values used to monetise changes in life-years lost were originally estimated by (Chilton et al, 2004). This study estimated a VOLY associated with a life-year spent in good health of £27,630 and in poor health of £14,280 (2002 prices). This was based on a survey of participants undertaken in between November 2002 and January 2003. Uplifts have been applied to ensure these values are relevant to the valuation of impacts today.

The air quality appraisal guidance recommends that all estimates of WTP to avoid detrimental health outcomes are uplifted annually by 2%. This advice reflected guidance published by the (Department of Health, 2004) and represents the view that willingness to pay (WTP) to avoid detrimental health effects is influenced by (and hence can be expected to rise in line with) the income of the person or household. For the updated damage costs, the original values from therefore needed to be updated for both real income growth from 2002 and price base (the price base for the updated damage costs is 2025).

Real income growth has been relatively low over the period since 2002. Hence it was considered inappropriate to use a fixed 2% uplift each year to represent real income growth. Instead, data for real GDP per capita were sourced from the Transport Analysis Guidance (TAG) (DfT, 2025) to derive a trend for real income growth. Rather than growing at an assumed 2% per annum, these data suggested instead that real incomes on average have only increased at an average rate of around 0.75% per annum from 2002 to 2025. (Unlike the previous report which used a simple average rate, this calculation is based on the compound annual growth rate.) Hence, using the assumed uplift could have led to substantial overestimation of the value of impacts.

From the 2023 iteration onwards the approach to assuming a proxy for income growth based on the long-run rate of economic growth per annum has been removed as the updated Green Book discount rates already consider the 2% uplift. The price base of the VOLY estimates was updated using the latest set of GDP deflators published by (HMT, 2025).

3.7.2 Morbidity associated with short-term exposure (Hospital admissions)

The original unit values used in the first estimation of damage costs have been uplifted using the latest data on real GDP per capita growth. These unit values capture both the disutility and resource cost associated with a hospital admission of each type, with the range representing uncertainty in monetary estimates of disutility.

3.7.3 Morbidity associated with long-term exposure (QALYs)

The damage costs include a number of morbidity pathways associated with long-term air pollution exposure, based on the approach taken by PHE. To value these longer-lasting morbidity impacts, health outcomes in terms of changes in incidence are converted into QALYs, which are then combined with a consistent unit monetary value of a QALY.

For the current update, the monetary WTP value for a QALY in the workbook has been updated to £84,548 in 2025 prices, which reflects the latest Green Book guidance with an adjustment to the price year. (HM Treasury, 2024).

The calculation of QALY loss requires utility weights for the different diseases, which are then multiplied by the change in incidence as in Equation 3.7.3.1:

Equation 3.7.3.1

$$QALY\ Loss_i = (1 - w_i) \cdot \delta_i \cdot \Delta I_i$$

Where:

$QALY\ Loss_i$ are the quality-adjusted life years for disease *i*.

w_i is the utility weight for disease *i*.

δ_i is the discounted duration of disease *i*.

These weights represent the QALY loss associated with each condition whilst living with the condition.

The utility weights for the 2023 damage costs were taken from (Sullivan, P. et al., 2011) Catalogue of EQ-5D scores for the United Kingdom. Males and females were allocated the same EQ-5D score and the diseases were mapped onto conditions listed in the publication using matching, or closest matching ICD-9 Categories. A QALY loss estimate for chronic bronchitis was taken from Solomon et al (2012), as discussed in (COMEAP, 2016). The set of utility weights used are presented in the following Table 3-8, alongside their source.

Table 3-8 List of EQ-5D values (QALY weights) allocated to males and females for each disease

Disease	w_i (2020)	Mapped ICD-9 Categories / description of disease (2020)
Chronic bronchitis	0.768	Weighted average of QALYs between COPD (moderate) and COPD (Severe), weighted according to effects of age and disease status on quality of life (1). Used to assess chronic phlegm
Asthma	0.722	ICD-9 493 Asthma (2)
CHD / IHD	0.61	ICD-9 410 Acute Myocardial Infarct (2)
Stroke	0.63	ICD-9 433 Precerebral Occlusion (2)
Diabetes	0.66	ICD-9 250 Diabetes Mellitus (2)
Lung cancer	0.56	ICD-9 162 Malignant Neoplasm Trachea/Lung (2)

(1) Salomon et al (2012) and P Burney as presented in (COMEAP, 2016); (2) (Sullivan, P. et al., 2011);

The duration of the disease is reflected in the δ_i, which is calculated according to Equation 3.7.3.2:

Equation 3.7.3.2

$$\delta_i = 1 \qquad \text{if } D = 1$$

$$\delta_i = 1 + \sum_{j=2}^{D} (1+r)^{1-j} \qquad \text{if } D > 1$$

Where:

D is the average years of duration of the disease.

r is the discount rate (referenced in Section 3.7.4)

The average years of duration of the disease were provided by Defra and were calculated using the DISMOD II model (WHO, 2018) and estimated based on the years of life with disability (YLD). The specific average years of duration for the diseases in this study are presented in Table 3-9. As the duration of the disease has been taken into consideration, the QALY loss (which, by definition, looks at the impact of living with the condition for a single year) can provide an indication on the lasting effects that conditions have beyond the first year.

Table 3-9 Average and discounted duration of disease

Disease	D [years]	δ [years]
IHD (angina)	9.50	8.92
IHD (AMI)	1	1
Asthma in Adults	23.60	20.05
Asthma in Children	36.20	28.22
Stroke	14.80	13.38
Diabetes	9.10	8.57
Lung cancer	1.80	1.79

By combining the change in incidence, with the QALY weight of living one year with the disease, and the (discounted) duration of the disease, this then calculates the cumulative QALY weight over the expected duration of the diseases associated with all incidences of the disease in a given year.

Finally, the costs produced by increases in the concentration of either $PM_{2.5}$ or NO_2 is the product of the valuation of a QALY loss and the quality-adjusted life years for disease *i* as in Equation 3.7.3.3 (see Table 3-11):

Equation 3.7.3.3

$$Cost_i = QALY\ Value \cdot QALY\ Loss_i$$

Following discussion with PHE regarding the strength of the underpinning epidemiological evidence, these long-term morbidity pathways were included in the damage costs (and the sensitivity around central values) as set out in Table 3-10. No pathways are included in the low damage cost.

Table 3-10 Inclusion of PHE pathways in damage cost sensitivities

	Long term exposure to $PM_{2.5}$	Long term exposure to NO_2
Low Damage cost	-*	-*
Central damage cost **(Stronger evidence suggestive for a causal association)**	Coronary heart disease Stroke Lung cancer Asthma (children)	Asthma (children)
High damage cost **(Evidence less certain or emerging evidence of associations)**	Chronic Obstructive Pulmonary Disease (as chronic bronchitis) Diabetes	Asthma (adults) Diabetes Lung cancer

* No pathways should be included in the low damage cost.

3.7.4 Discounting

This damage or impact from air pollution (or exposure to air pollution) is observed over a long period of time. That is, if you are exposed to pollution today, the health impacts will be spread over your lifetime and not condensed into a single year. The damage cost values, therefore, illustrate the present value of the impacts from air pollutant emissions over a period. To do so, discounting is required.

Discounting is 'based on the concept of time preference, which is that generally people prefer value now rather than later', and 'converts costs and benefits [or impacts] into present values by allowing for society's preference for now compared with the future' (HMT, 2024). Discounting is thus applied in this case to express intertemporal impacts of air pollutant emissions in a given year as a present monetary value. For illustration purposes, the following paragraphs set out the steps taken to calculate the monetised damage costs from mortality attributable to $PM_{2.5}$ exposure (as the most significant health impact pathway).

Firstly, the impacts of exposure to air pollution concentration on people's mortality risks are estimated. Although the damage cost values are presented in terms of £/tonne this step is based the change in pollutant concentrations and the application of the relevant concentration response functions (as provided by the Committee on the Medical Effects of Air Pollution (COMEAP))[12]. In this case, the health impacts are calculated as Life Years Lost (LYL) resulting from human exposure to $PM_{2.5}$. This metric reflects the total number of life years lost across a population (rather than a single individual) and aggregates this across the exposed population.

The LYL are attributable to air pollutant exposure only (i.e. pollutant exposure is not the only determinant) and considers the cumulative health burden where small reductions in life expectancy across the population (inc. less than a year) are aggregated. As an example, if air pollution exposure led to a 1-day decease in life expectancy for 365 people, this would be considered 1 full LYL. As the health impacts from exposure are spread over a period of time, the LYL are calculated over a 100-year period, illustrating the impact within each given year resulting from exposure in year one. The estimation of these impacts is informed

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[12] The full detailed approach to the estimation of health impacts is detailed in the accompanying damage cost reports https://uk-air.defra.gov.uk/assets/documents/reports/cat09/2301090900_Damage_cost_update_2023_Final.pdf

using life tables, which are applied to calculate the mortality effects associated with long-term exposure to $PM_{2.5}$ and NO_2. An overview of the methodology and application of the life tables can be found in Section 3.4.

Secondly, the impact in terms of LYLs is valued economically. As noted previously, the social value of one LYL, in real monetary terms, used in Defra's damage cost work so far is based on a study originally conducted by (Chilton et al, 2004). This study estimated a VOLY associated with a life-year spent in good health of £27,630 and in poor health of £14,280 (2002 prices). This was based on a survey of participants undertaken in between November 2002 and January 2003. This has been adjusted to 2024/5 prices in the current updates to the damage cost analysis workbook.

Thirdly, the next step is to apply a discounting approach, in line with the UK Green Book, to account for social time preference and develop an estimate of the present value of the damage costs from mortality attributable to $PM_{2.5}$ air pollution. In particular, the impacts are discounted using the UK Green Book; that is, discount rate (for human health effects) of 1.5% for year 0 to 30, 1.29% for year 31 to 75 and 1.07% for year 75 to 125 (HMT, 2024). This enables a single present value to be calculated taking into account the discounted values of lifetime health impacts associated with exposure to $PM_{2.5}$ (in this example).

Discounting is only applied to mortality impacts associated with long-term exposure and morbidity effects assessed through changes in incidence. Discounting is not applied to other impact pathways in which the impacts occur in the year when the change of emissions occur. Specifically:

1. Productivity impacts: these impacts are represented by a change in WDL and mRADs, which are acute events and happen fairly shortly after exposure to changes in pollution. These are assumed to occur in year[13].
2. Material damage and building soiling: the value of the damage estimate has been annualised and can, therefore, be treated as if the impacts occur in year.
3. Ecosystems: these values are taken from an underlying study which recommends damage costs for appraisal. Hence, impacts are either in year or already discounted in the recommended values.

3.7.5 Summary of unit values for the valuation of health outcomes

The unit values used in the valuation of health outcomes are set out in Table 3-11 below. The 2025 damage cost update has not used Monte Carlo analysis to derive central estimates of the damage costs within sensitivity bounds. Instead, an average of the 'high' and 'low' bounds for the value of hospital admissions is taken to provide a central estimate of the cost. This results in a central cost of £11,200 and £11,400 for respiratory and cardiovascular hospital admissions respectively.

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[13] It is assumed that there will be no long-term impacts on productivity.

Table 3-11 Unit values used in health impact valuation*

Health effect	Metric description	Unit values used in valuation in 2025 Update (£2025 prices) (Sensitivity range)	Unit values used in valuation in 2023 update (£2022 prices)** (Sensitivity range)
Mortality attributable to short-term exposure	Number of years of life lost due to air pollution, assuming 2-6 months loss of life expectancy for every death brought forward. Life expectancy losses assumed to be in poor health	£30,600 (10-15% of LYL valued using 'good health' VOLY)	£26,200 (10-15% of LYL valued using 'good health' VOLY)
Mortality (long-term exposure)	Number of years of life lost due to air pollution. Life expectancy losses assumed to be in normal health.	£59,200 (£37,400 – 73,800)	£50,600 (£37,900 – £63,100)
Respiratory hospital admissions	Case of a hospital admission, of average duration 8 days	£11,200 (£3,800 – £18,600)	£9,800 (£3,300 – £16,300)
Cardiovascular hospital admissions	Case of a hospital admission, of average duration 9 days	£11,400 (£4,000 – £18,800)	£10,000 (£3,500 – £16,500)
QALY loss	Cumulative discounted QALYs over duration of disease	£84,500 (£42,300 - £112,700)	£72,000 (£36,000 - £96,100)

*Values rounded to nearest £100

** (Defra, 2023)

4 Estimation and valuation of non-health impact pathways

4.1 Material damage and building soiling

For the 2025 damage cost update, the general approach to estimating material damage and building soiling is largely unchanged from the methods applied in the 2023 damage cost update.

Three pathways have been included in the damage cost estimates. The pathways are limited to those where air pollution degrades or soils materials and buildings. Given the scope of the project, the cost estimates have been adapted from the original damage cost calculation rather than being re-estimated.

Concentrations of air pollutants in the atmosphere have been proven to have a detrimental impact on buildings in utilitarian applications (i.e. in houses, factories, etc.). The quantification of these impacts was assessed within various studies for the European Commission DG Research, in particular ExternE and associated projects. The pollutants most implicated in acid damage are SO_2 (most importantly), H+ and NO_2. The most significant impacts are on natural stone and zinc coated materials. The benefits of reducing material damage from SO_2 have been included in the updated damage cost estimates using the methodology used for the original damage costs (although with an update to the price base): this suggested an impact of around £316 per tonne of SO_2 emitted (2025 prices).

Damage to building materials covers limestone, sandstone, mortar and zinc used in galvanised steel. Quantification covers utilitarian buildings and infrastructure, but not cultural heritage. Response functions were taken from a major international research effort and are based on 8 years of exposure of material specimens across Europe. These demonstrate SO_2 to be the most harmful of the pollutants under conditions up to the mid-2000s, so analysis has focused on this pollutant. Valuation is performed using repair cost data from the architecture and building sector, with repair assumed necessary once a critical loss of material (defined in relation to each material, taking account of how they are used) has occurred. Value is calculated via the change in frequency of repair operations. Full account of the methods used is provided in the reports of the European Commission funded ExternE Project (ExternE, 1995, p. 300; ExternE, 1998, p. 381; ExternE, 2005, p. 109).

Ozone can also have a damaging impact on materials, in particular on rubbers and paints exposed to ambient air. (Holland, M. et al, 1998) undertook a large study into the impacts of ozone on a range of paints and rubber formulations representative of those in the UK market. The study found that impacts on paint were unlikely over the lifetime of their application but did quantify a relationship between ozone and damage to rubber materials. The effect of a population weighted 1ppb change in ozone was estimated at £3.7m per annum (2005 prices). This relationship has been used in the new damage cost estimates with an update to the price base and conversion to be expressed in terms of population-weighted ozone concentration (1ppb to $2\mu gm^{-3}$) to gain the impact per tonne of NO_x or VOC emitted via this ozone pathway in 2025 prices.

Soiling of buildings by particles is one of the most obvious signs of pollution in urban areas. The degree of soiling of particles varies according to a number of factors specific to the particles themselves, the nature of emission, the surface affected and wider meteorological conditions: for example, blackness per unit mass of smoke, particle size distribution, and chemical nature of the particles. Although the relationship between particle emission and soiling is strong, quantification of impacts is not straightforward. The original damage cost estimates used an approach developed by (Rabl, Curtiss, & Pons, 1998) which captured both the cleaning and amenity costs associated with building soiling. The same approach is adopted here which suggests that a 1 tonne change in PM_{10} has an associated cost of £751 (2025 prices).

In the 2020 damage costs, in contrast to the other $PM_{2.5}$ pathways, the damage cost for building soiling does not take the location of emissions, dispersion conditions or the density of stock at risk into account. This resulted in the contribution from this $PM_{2.5}$ pathway relative to other pathways is relatively higher for some sector damage costs (e.g. part A sectors and for agriculture) relative to others. However, this result appears inconsistent with the logic that building soiling impacts are likely to be higher in urban centres where there is a greater density of buildings (and people), relative to industrial or agricultural settings where the density of buildings (and/or the amenity value attached to the appearance of such buildings) is likely to be lower. As such, for the 2023 update, the approach to estimating building soiling effects has been adjusted such that the effects scale with human health impacts (implicitly assuming that the density of buildings – or the density of buildings to which an amenity value is attached – is correlated with the density of population).

4.2 Ecosystem impacts

For the 2025 damage cost update, the general approach to estimating ecosystem impacts is largely unchanged from the methods applied in the 2023 damage cost update.

As part of the scoping exercise, Ricardo has explored the guidance and literature concerning the valuation of ecosystem impacts, with initial findings demonstrating a growing evidence base within this area.

Following discussion with the IGCB, it was decided that the ecosystem pathway evaluation approach used previously would be retained within the 2025 update, but the relationship between air pollution and ecosystems would be continually monitored with a view to further updates being included in future damage cost iterations.

4.2.1 Overview of ecosystem pathway valuation

A key gap in the quantification of impacts associated with changes in air pollution are the effects on environmental health and the services ecosystems provide. The strength of evidence and methodologies to quantify these effects has lagged that of human health effects given the latter have been prioritised over the last couple of decades. That said, the initial set of damage costs did include impacts on crop yields. To start addressing this gap, Defra commissioned a tranche of projects to explore the impacts of air pollution on ecosystem service provision. One of the outputs of this work was a report by Jones et al. titled 'Assessment of the Impacts of Air Pollution on Ecosystem Services – Gap Filling and Research Recommendations' (Jones, Mills, & Milne, 2014). The aims of this study were to:

1. Review the evidence and data behind previous valuation studies of air pollution on ecosystem services.
2. Apply an improved spatially explicit methodology to value impact of selected ecosystem services.
3. Prioritise additional ecosystem services for valuation of air pollution impacts. Identify existing or planned projects and new research which might provide relevant information and recommend appropriate research approaches to model them.
4. Collate damage costs from this and previous studies.

The study reviewed the evidence linking air pollution to a range of potential impacts on ecosystem services and collated damage costs associated with several pathways.

Alongside collating the damage costs, Jones et al. (2014) provided direction on the rigour of the value estimate (Jones, Mills, & Milne, 2014). To do so they scored each damage cost as either '## Robust', '# Acceptable' or '(#) Improvements desirable and not currently acceptable for policy appraisal'. As per the 2023 update, only the 'robust' pathways are included in the damage costs. As a consequence, only the detrimental impact of additional N on biodiversity is included associated with additional N for the NOx and NH_3 damage costs.

Jones et al. (2014) also provided uncertainty ranges around the valuation of each damage (Jones, Mills, & Milne, 2014). Following steer from IGCB, for those pathways included based on the rigour of the estimate, the low valuation sensitivity is included in the low damage cost, the central in the central damage cost and the high in the high damage cost. The pathways included in the updated damage costs and the sensitivity range around the central valuation are presented in Table 4-1.

Table 4-1 Ecosystem service impacts included in the updated damage costs based on Jones et al (2014)

Pollutant	Unit	Sensitivity	Provisioning services			Regulating services			Cultural services	
			Crop production	Timber production	Livestock production	CO_2 GHG Emissions	N_2O GHG Emissions	CH_4 GHG Emissions	Recreational fishing	Biodiversity
NO_2	£/tonne (2014 prices)	Central	-	-4.30	-8.80	-54.00	11.80	-	0.10	102.80
		Low	-	*-2.30*	*-5.60*	*-22.80*	*6.20*	-	*0.10*	*33.30*
		High	-	*-8.00*	*-11.80*	*-94.00*	*18.70*	-	*0.10*	*237.40*
NH_3	£/tonne (2014 prices)	Central	-	-93.10	-294.10	-1,267.10	338.40	-	2.20	413.80
		Low	-	*-49.70*	*-186.60*	*-535.40*	*179.10*	-	*2.20*	*139.10*
		High	-	*-170.70*	*-395.90*	*-2,204.00*	*537.40*	-	*2.20*	*1,021.50*
SO_2	£/tonne (2014 prices)	Central	-	-	-	-	-	-5.30	-	-
		Low	-	-	-	-	-	*-1.60*	-	-
		High	-	-	-	-	-	*-9.50*	-	-
O_3^*	£/ppb (7-month 24-hour mean) (2014 prices)	Central	-	-	1,051,000	5,740,000	-	-	-	-
		Low	-	-	*427,000*	*3,866,000*	-	-	-	-
		High	-	-	*1,705,000*	*7,939,000*	-	-	-	-
O_3^*	£/POD (2014 prices)	Central	100,555,000	-	-	-	-	-	-	-
		Low	*83,421,000*	-	-	-	-	-	-	-
		High	*118,970,000*	-	-	-	-	-	-	-

‘-‘ denotes no relevant impact / no impact assessed, * (Jones, Mills, & Milne, 2014) present costs as a negative integer and benefits as a positive integer for decreases in NO_2, NH_3 and SO_2 emissions and increases in O_3 metrics. This table presents costs as positive integers, associated with an additional unit of pollution (to be consistent with the way damage costs are presented in the rest of the report). As such we have reversed the sign of the values for O_3 impacts so that costs are shown as a positive integer associated with a unit increase for all pollutants (-ve numbers are benefits associated with an increase in emission).

Ref: Ricardo/ED12943/

4.2.2 Valuation Approach

The valuation approach for ecosystems services has remained consistent with the previous 2023 damage cost update. This is described below for completion.

4.2.2.1 Nitrogen impacts on livestock production

The report by Jones et al. (2014) analysed the impact of nitrogen deposition on improved grassland (Jones, Mills, & Milne, 2014). The estimations are reliant on the assumption that farmers observe the effects of changes in nitrogen input from atmospheric deposition and offset this by varying fertiliser application. The impact of air pollution on livestock production is assessed via the effect on the productivity of grassland, and consequently meat (cattle and sheep) and dairy production. Potential air pollution impacts are through the effect of: (i) nitrogen deposition on grassland productivity; and (ii) ozone on grassland productivity.

Following this assumption, increased deposition of nitrogen increases the growth rate of improved grassland habitats. This reduces the needs for application of nitrogen fertilisers for the management of the land, and results in lower production input costs to farmers. Decreases in nitrogen deposition are assumed to have the opposite effect. Give the lack of information available to quantitatively link ozone to changes in livestock production, the analysis focuses on the impact of nitrogen deposition on improved grassland.

Farm gross margin (FGM) (£ per hectare or £ per head) is a widely used measure of the value of different agricultural land uses and enterprises. It is defined as the difference between revenues from agricultural activities and associated variable costs, i.e. the value of crop and livestock output minus variable costs[14]. Output includes the market value of production that is retained by farmers. Variable costs for grazing livestock include feed and forage crop (the cost of which is includes fertiliser, seed and sprays) (Redman, 2011). Since FGM excludes fixed capital costs and is estimated net of variable input costs, it could also be considered to provide an approximation of the ‘ecosystem service value’ associated with the provision of livestock. The change in FGM associated with livestock production is estimated in terms of the change in nitrogen fertiliser input cost. The John Nix Farm Management pocketbook reports indicative fertiliser input prices in the UK, based on August 2010 spot prices (Redman, 2011). Based on the content in the straights, the pocketbook calculates the average price of nitrogen in fertiliser to be £0.62 per kg. This value was applied in the 2020 damage cost update.

Market prices for ‘straights’ which contain nitrogen for plant stem and leaf growth have increased greatly since 2011 and are reported in the 2019 John Nix Pocketbook for Farm Management (Redman, 2019):

- Ammonium nitrate (34.5% N): £216 - 225 per tonne
- NS grade (27% N, 30% SO_3): £224 per tonne
- Sulphate of ammonia (21% N, 60% SO_3): £232 per tonne
- Urea (46% N): £300 per tonne
- Liquid nitrogen (26% N, 5% SO_3): £177 per tonne.

These indicative fertiliser input prices are based on forward prices in October 2017 and spring 2019. Based on the content in the straights, the pocketbook calculates the average price of nitrogen in fertiliser to be £0.65 per kg. This value is applied in the analysis. The ecosystem service value used for the estimation of the updated damage costs are set out in Table 4-2.

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[14] Fixed costs (rent, labour, machinery and general overheads) are excluded from FGM since these have to be covered across all farm activities.

Table 4-2 Ecosystem service value applied in updated damage costs (£2025 per tonne) - Nitrogen impacts on livestock production

Pollutant	Unit	Sensitivity	Livestock production
NO_2	£/tonne (2025 prices)	Central	-12
		Low	*-7*
		High	*-16*
NH_3	*£/tonne (2025 prices)*	Central	*-392*
		Low	*-249*
		High	*-527*

4.2.2.2 Ozone impacts on livestock production

Ozone decreases the forage quality and the yield of pasture, which causes decreases in lamb growth if pasture is the sole food source. Desired lamb growth rates can be obtained with poorer quality pasture if sufficient supplementary feed is given. Ozone impacts on lamb were valued in terms of changes in the amount of concentrate required to get lambs to target weight.

Lambs can be finished on a mixture of forage and silage or grain, but the proportions vary according to individual farmers. The previous versions of the air quality damage cost used four suggested formulations for concentrate feed for lambs, with associated prices per tonne (assuming the farmer is mixing it themselves, and no cost for mixing is included) from (Eblex, 2009). The mean value of these four prices was used (£176.50 per tonne; from range £167-£188). The same four suggested formulations for concentrate feed for lambs were adopted and updated by using more recent cost concentrate prices from the 2019 John Nix Pocketbook for Farm Management (Redman, 2019).

Table 4-3 Ecosystem service value applied in updated damage costs (£2025 per ppb) – Ozone impacts on livestock production

Pollutant	Unit	Sensitivity	Livestock production
O_3	£/ppb (2025 prices)	Central	-1,518,839
		Low	*-617,074*
		High	*-2,463,959*

4.2.2.3 Net GHG emissions

Values for non-traded GHG emissions are applied since the analysis focuses on the GHG regulation service of woodland, heathland and other semi-natural habitats and the impact that air pollution (nitrogen, sulphur and ozone) has on the net emissions of GHGs – specifically carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) - from these habitats. The nitrogen, sulphur and ozone impact pathways are described below.

Nitrogen impacts on GHG emissions

- Nitrogen deposition impacts the sequestration of CO_2 through changes in plant and tree growth, and subsequent long-term storage of carbon in soils. Increased deposition results in increased potential for growth and hence increased sequestration. Decreased deposition, i.e. recovery, has the opposite effect.

- Nitrogen deposition impacts on N_2O emissions, whereby a proportion of nitrogen deposition is re-emitted into the atmosphere.

Sulphur impacts and acidity impacts on GHG emissions

- Sulphur deposition as a nutrient has also been found to impact on plant and tree growth although to a less significant extent than nitrogen.

- Sulphur deposition potentially influences N_2O emissions through acidification of soil. However, for agricultural habitats it is assumed that soil acidity is controlled by practices such as liming; hence changes in sulphur deposition are offset by agricultural land use management.

- Sulphur deposition has been found to reduce net CH_4 emissions from wetlands and bogs.

Ozone impacts on GHG emissions

Ozone impacts sequestration of CO_2 in woodlands and grasslands through biomass reduction due to ozone damage at accumulated ozone doses above a threshold ozone concentration of 40 ppm (AOT40).

The non-traded price of GHG emissions of £55/tonneCO_2e for 2014 was sourced from (DECC, 2011). The non-traded carbon price from the Green Book supplementary guidance of £273/tonneCO_2e for 2025 was applied in this analysis to scale up the costs (DESNZ, 2023).

Table 4-4 Ecosystem service value applied in updated damage costs (£2025 per tonne) – Nitrogen and ozone impacts on CO_2 emissions

Pollutant	Unit	Sensitivity	CO_2 GHG
NO_2	£/tonne (2025 prices)	Central	*-234*
		Low	*-99*
		High	*-407*
NH_3	*£/tonne (2025 prices)*	Central	*-5,489*
		Low	*-2,319*
		High	*-9,547*
O_3	*£/tonne (2025 prices)*	Central	*-24,864,689*
		Low	*-16,746,844*
		High	*-34,390,377*

Table 4-5 Ecosystem service value applied in updated damage costs (£2025 per tonne) – Nitrogen impacts on N2O emissions

Pollutant	Unit	Sensitivity	N_2O GHG
NO_2	£/tonne (2025 prices)	Central	*51*
		Low	*27*
		High	*81*
NH_3	*£/tonne (2025 prices)*	Central	*1,466*
		Low	*776*
		High	*2,328*

Table 4-6 Ecosystem service value applied in updated damage costs (£2025 per tonne) – Sulphur impacts on CH4 emissions

Pollutant	Unit	Sensitivity	CH_4 GHG
SO_2	£/tonne (2025 prices)	Central	-23
		Low	*-7*
		High	*-41*

4.2.3 Updates to valuation approach in 2025 update

4.2.3.1 Ozone impacts on crop production

Defra commissioned a report by Jones et al. (2014) that analysed the ozone impacts on wheat production (Jones, Mills, & Milne, 2014). Ozone impacts on wheat were calculated only for the future scenario, using the ozone flux metric of Phytotoxic Ozone Dose above a threshold of 6 nmol m^{-2} s^{-1}(POD_6) at 10 x10 km resolution.

Under a future ozone scenario, the loss of production due to ozone replicates the spatial pattern of current wheat production, with ozone fluxes being highest in those areas where wheat is extensively grown. Impacts of ozone on wheat production were calculated using the spatially explicit change in yield and therefore production, coupled with the value transfer evidence. Calculation of economic loss used the five-year average farm gate wheat value, centred on 2007 (£109/tonne).

In the 2025 update, the value transfer guidance has been updated to reflect a more recent five-year average farm gate wheat value estimate, centred on 2022. The average agricultural price index (API) is a set of indices of the prices paid and received by UK farmers for agricultural goods and services. The average API of wheat for the period 2005-2009 (84, 2015=100) and 2020-2024 (125, 2020=100) was taken from Defra’s latest API (Defra, 2025). The 107% change in API was then used to adjust the five-year average farm gate wheat value, centred on 2007 to 2022 (£225/tonne). The ecosystem service impacts of crop production were increased by 107% in the analysis to reflect the most recent valuation of wheat. The ecosystem service value used for the estimation of the updated damage costs are set out in Table 4-7.

Table 4-7 Ecosystem service value applied in updated damage costs (£2025 per tonne) - Ozone impacts on wheat production

Pollutant	Unit	Sensitivity	Crop production
O_3^*	£/POD (2025 prices)	Central	*-238,505,661*
		Low	*-197,865,653*
		High	*-282,184,063*

5 Activity costs and other updates

5.1 Background

Air pollutant activity costs, like damage costs, summarise a valued impact of air pollution in a form for ready application in appraisal. Activity costs present the impact of air pollution per unit of energy consumed, rather than per tonne of pollutant emitted (as is the case with damage costs). As such, activity costs can be used where changes in emissions arising from a policy are unknown, preventing the application of the damage costs.

The activity costs are published as part of BEIS's Green Book supplementary appraisal guidance. Given they are derived using IGCB's air pollutant damage costs, these values capture the same subset of air pollutant impacts.

Activity costs are differentiated by location of fuel use to capture the differential in policy impact of reducing air pollutant emissions from fuel use between, for example inner city, where impacts will be higher, and rural areas. Table 5-1 below presents a breakdown of what is covered by the updated activity costs and the corresponding results are presented in section 8.

Table 5-1 Breakdown of updated activity costs produced

Activity cost set	Area type	Fuel split
National average	N/A	Electricity, gas, coal, oil, biomass, LPG, peat, petroleum coke
Domestic	Inner conurbation, Urban big, Urban medium, Urban small, Rural	Gas, coal, oil, biomass, LPG, peat, petroleum coke
Transport	Transport average, Central London, Inner London, Outer London, Inner conurbation, Outer conurbation, Urban big, Urban large, Urban medium, Urban small, Transport rural	Car petrol, car diesel, LGV petrol, LGV diesel, Rigid HGV diesel, Articulated HGV diesel

As part of the damage cost update 2025, an updated set of activity costs have also been produced (to carry through the underlying changes to the damage costs themselves). In addition, specific updates to the activity costs have been made (as set out in further detail below), including adopting the latest NAEI 2022 emission factors and updating price base to 2025.

5.2 Methodology

5.2.1 Overview

To be able to specify the activity costs per unit of fuel consumed, data were taken from the National Atmospheric Emissions Inventory (NAEI) to define an effective air pollutant emissions factor for the different fuels under consideration (quantity of emissions per unit of fuel used). These emission factors were then combined with the relevant air pollutant damage cost to capture and value air pollutant impacts per unit of fuel consumption. The calculation used the 2025 updated set of air pollutant damage costs.

The 2025 damage costs were derived from air pollutant modelling that was calculated using emission estimates from the NAEI 2022. Hence emission factors were also derived from NAEI 2022.

The methodology used in each case is as follows:

Transport activity costs

1. Take UK total $PM_{2.5}$, NOx and SO_2 emissions from road traffic in 2022 from NAEI 2022, split by vehicle type and fuel

2. Take UK total fuel consumption for road traffic in 2022 by vehicle type and fuel from NAEI 2022
3. Divide emissions by fuel consumption to calculate an emissions factor for each vehicle type and fuel
4. Combine the emissions factors with $PM_{2.5}$ and NOx transport area-specific and SO_2 national damage costs to calculate activity costs.

National average (primary fuel) activity costs

1. Take UK total $PM_{2.5}$, SO_2 and NOx emissions from NAEI split by fuel (gas, coal, oil, biomass, LPG, peat, and petroleum coke) in 2022 for all sectors
2. Take UK total fuel consumption (gas, coal, oil, biomass, LPG, peat, and petroleum coke) in 2022 from NAEI for all sectors
3. Divide emissions by fuel consumption to calculate an average emissions factor for each fuel
4. Combine the emissions factors with national average $PM_{2.5}$, SO_2 and NOx damage costs to calculate activity costs.

Domestic Activity Costs split

1. Take UK total $PM_{2.5}$, SO_2 and NOx emissions from NAEI split by fuel (gas, coal, oil, biomass, LPG, Peat, and Petroleum Coke) in 2022 for the domestic sector
2. Take UK total fuel consumption (gas, coal, oil, biomass, LPG, Peat, and Petroleum Coke) in 2022 from NAEI for the domestic sector
3. Divide emissions by fuel consumption to calculate an emissions factor for domestic emissions for each fuel
4. Combine the emissions factors with new $PM_{2.5}$ and NOx proxy damage costs for the domestic sector, split by area type and fuel type (see 'Emissions to Concentrations modelling' section below)
5. Combine SO_2 emission factors for the domestic sector with national average damage costs for SO_2
6. Calculate the activity costs for the domestic sector as the sum of the activity costs for $PM_{2.5}$, SO_2 and NOx.

Activity costs associated with electricity consumption

1. Take 2025 damage costs for $PM_{2.5}$ Part A Category 5, NO_X Part A Category 5 and SO_2 National emissions from the damage cost workbook
2. Take historic 2022 emissions of NOx, $PM_{2.5}$ and SO_2 from NAEI 2022 for 'power stations'
3. Take final consumption (excluding international aviation) of electricity for 2022 from DUKES
4. Divide emissions by final consumption of electricity to calculate an average emissions factor for NOx, $PM_{2.5}$ and SO_2 for 2022.
5. Project forward the emissions factor, using the consumption-based 'Grid average' emissions factors from BEIS Supplementary Green Book guidance. Convert to a proxy Long-run marginal consumption-based emissions factor by applying the ratio between grid-based and long-run marginal CO_2 factors in each year from BEIS' guidance (this step implicitly assumes the trend for air pollutant emissions intensity will move in line with the GHG emissions intensity of the marginal plant).
6. Combine the emissions factors with damage costs for $PM_{2.5}$ Part A Category 5, NOx Part A Category 5 and SO_2 National emissions in each projected year.
7. Calculate the total activity cost as the sum of the activity costs for $PM_{2.5}$, SO_2 and NOx.

The approach to estimating the above activity costs is consistent with that used for the 2023 update.

Where a profile of activity costs is defined over time (i.e. for national primary fuel, domestic primary fuel, and electricity consumption), a 2% per annum uplift to reflect rising income over time is not applied to define the activity costs in future years. This reflects HMT Green Book guidance. This in mind, analysts should therefore discount any air pollution impacts calculated using the activity costs using the HMT Green Book discount rate for risk to health and life values of 1.5%.

5.2.2 Emissions factors

The methodology uses a range of data regarding emissions and fuel use taken from NAEI for 2022. The 2022 NAEI was used within the PCM modelling that, in turn, was used to derive the updated damage costs on which these activity costs are based. In order to be consistent with the emission inventory assumptions, including emission factors, the methodology extracted the various emissions totals and activity totals from the 2022 NAEI in order to calculate the implied emission factors in terms of emission per unit of activity (per kWh or per litre) needed for the activity costs. The emissions and fuel data extracted and emissions factors calculated are included in the table below.

Table 5-2 Emissions and fuel data used and emissions factors calculated

Sector	Sub-sector	Total emissions in 2022			Total fuel consumption in 2022	Calculated emissions factor		
		$PM_{2.5}$	NOx	SO_2		$PM_{2.5}$	NOx	SO_2
			kilotonnes		*Million litres*		*g/litre*	
Transport	Car petrol	0.145	18.026	0.095	13,502	0.011	1.335	0.007
	Car diesel	0.966	86.389	0.128	10,494	0.092	8.232	0.012
	LGV petrol	0.002	0.327	0.002	213	0.010	1.530	0.007
	LGV diesel	0.527	64.104	0.084	6,846	0.077	9.364	0.012
	Rigid HGV diesel	0.148	12.498	0.031	2,514	0.059	4.972	0.012
	Articulated HGV diesel	0.093	6.403	0.057	4,638	0.020	1.380	0.012
			kilotonnes		*kWh*		*tonnes/GWh*	
National	Gas	0.749	121.465	1.932	705,203,146,075	0.001	0.172	0.003
	Coal	4.388	13.390	35.719	32,139,097,113	0.137	0.417	1.111
	Oil	4.619	152.398	6.769	103,670,136,227	0.045	1.470	0.065
	Biomass	18.590	24.219	2.945	85,628,918,323	0.217	0.283	0.034
	LPG	0.047	5.356	0.030	14,584,459,954	0.003	0.367	0.002
	Peat	0.037	0.002	0.001	12,664,392	2.952	0.180	0.040
	Petroleum coke	0.686	4.823	35.666	10,790,412,923	0.064	0.447	3.305
Domestic	Gas	0.211	16.694	0.248	229,682,633,816	0.001	0.073	0.001
	Coal	3.399	2.230	15.037	6,117,718,110	0.556	0.365	2.458
	Oil	0.114	4.046	0.372	21,159,051,325	0.005	0.191	0.018
	Biomass	14.027	1.751	0.168	9,475,655,072	1.480	0.185	0.018
	LPG	0.002	0.283	0.003	2,585,516,787	0.001	0.109	0.001
	Peat	0.037	0.002	0.001	12,664,392	2.952	0.180	0.040
	Petroleum coke	0.294	0.452	18.877	1,252,727,522	0.235	0.361	15.069

Only one emissions factor for each pollutant was calculated for each fuel for the domestic and transport sectors: this does not vary across area types.

For electricity, as noted, a dynamic set of emissions factors are used, reflecting the marginal plant (i.e. the last -i.e. highest cost - plant dispatched to meet demand) on the grid. The emissions factors assumed for key years are presented in the following table. A linear trend in between these key years is applied.

Table 5-3 Emissions factors for electricity consumption (g/kWh)

Pollutant	2025	2030	2035	2040	2045	2050
$PM_{2.5}$	0.004	0.002	0.001	0.000*	0.000*	0.000*
NOx	0.258	0.114	0.030	0.008	0.002	0.002
SO_2	0.035	0.016	0.004	0.001	0.000*	0.000*

*Value is not 0 but very small so does not appear due to rounding.

5.2.3 Emissions to concentration modelling

PCM model runs were completed to derive the emissions to concentration relationships for domestic emissions in the required locations (i.e. ‘area types’) and for the relevant fuels.

Separating by fuel within each area type is important because not only do different fuels have different emissions intensities, but also the consumption of different fuels have different spatial distributions relative to centres of population. This has an impact on the level of exposure of the population to the emissions (and resulting concentrations) generated through the consumption of different fuels. Natural gas is the dominant domestic fuel in large towns and cities but is not available in some rural communities. Fuels such as coal and oil by contrast are less widely used in large towns and cities but can be important fuels in communities without a natural gas supply.

The PCM model was used to calculate the ambient PM concentrations associated with each domestic fuel in each area type and the population-weighted mean concentration was then calculated for each of these combinations. This was then divided by the emissions total for each fuel in each area type in order to calculate the ugm^{-3} per tonne emitted. These outputs were then combined with health impact and valuation data to produce a set of specific, proxy ‘damage costs’ for use in the calculation of the updated activity costs. This was done using a consistent methodology to that used to update the damage costs. They are described as proxy damage costs because whilst they have been calculated using the same methods as the core damage costs, these particular values are not published. These proxy damage costs are listed in Table 5-4 below.

Table 5-4 Proxy damage costs created by PCM modelling for activity costs

Area type				
Inner Conurbation	**Urban Big**	**Urban Medium**	**Urban Small**	**Rural**
Domestic Gas Inner Conurbation	Domestic Gas Urban Big	Domestic Gas Urban Medium	Domestic Gas Urban Small	Domestic Gas Rural
Domestic Coal Inner Conurbation	Domestic Coal Urban Big	Domestic Coal Urban Medium	Domestic Coal Urban Small	Domestic Coal Rural
Domestic Oil Inner Conurbation	Domestic Oil Urban Big	Domestic Oil Urban Medium	Domestic Oil Urban Small	Domestic Oil Rural
Domestic Biomass Inner Conurbation	Domestic Biomass Urban Big	Domestic Biomass Urban Medium	Domestic Biomass Urban Small	Domestic Biomass Rural
Domestic LPG Inner Conurbation	Domestic LPG Urban Big	Domestic LPG Urban Medium	Domestic LPG Urban Small	Domestic LPG Rural
Domestic Peat Inner Conurbation	Domestic Peat Urban Big	Domestic Peat Urban Medium	Domestic Peat Urban Small	Domestic Peat Rural
Domestic Petroleum Coke Inner Conurbation	Domestic Petroleum Coke Urban Big	Domestic Petroleum Coke Urban Medium	Domestic Petroleum Coke Urban Small	Domestic Petroleum Coke Rural

For each activity cost, an emissions factor and activity cost are first calculated for each relevant pollutant type. The impacts across different pollutants are then summed to form the final activity costs for each fuel and area type. The updated activity costs are presented in Section 8.

6 Damage cost sensitivities

6.1 Introduction

For the 2025 damage cost update, the general approach to determining a sensitivity range around the central damage costs is largely unchanged from the methods applied in the 2023 damage cost update.

6.2 Uncertainty in the estimation of damage costs

The estimation of the impacts of air pollution on both health and non-health pathways is inherently uncertain. The methodology for assessing the different impact pathways (which are subsequently aggregated to form the damage costs) is based on a number of assumptions around which there is a distribution of probable outcomes. The updated damage costs estimated under this project represent a best estimation of a ‘central’ damage cost estimate. However, there is uncertainty around: the emissions dispersion modelling, the interpretation of changes in air pollution concentrations into impacts and the valuation of those impacts. In this update, only one uncertainty range has been developed to reduce the complexity of the use and interpretation of the damage costs.

6.2.1 Concentration response functions and pathway inclusion

CRFs are varied between the low and high damage cost estimates. For those pathways included in the central damage cost using the central CRF value, these are captured in the low damage cost applying the lower bound and in the high damage cost using the high bound of the CRF range.

Some pathways are excluded altogether from the central and low damage costs and are only recommended for inclusion in the high damage cost (e.g. chronic bronchitis). Where this is the case, the pathways are only included in the high damage cost based on the central value of the CRF range.

In the initial damage costs COMEAP (and subsequently IGCB) recommended a relationship between NO_2 and respiratory hospital admissions for quantitative analysis but noted that any impact should only be included as a sensitivity. COMEAP has noted that recent evidence including the REVIHAAP review (WHO, 2013), the HRAPIE project (WHO, 2013), the SGUL review itself (Mills, Atkinson, Kang, Walton, & Anderson, 2015), the SGUL adjustment for PM mass in two-pollutant models (Mills, Atkinson, Anderson, Maynard, & Strachan, 2016) , and current USEPA Integrated Science Assessments (USEPA, 2016; USEPA, 2019) suggests a causal role for NO2 in respiratory effects has strengthened in recent years. As such, this project includes this impact pathway in the low, central and high sensitivity damage costs.

A mapping of the point on the CRF range for each impact pathway across each damage cost is presented in Table 6-1.

For the effects of NO_2 on mortality, the sensitivity range also varies the adjustment applied to the CRF. This adjustment is applied to account for the overlap between the mortality impacts of NO_2 and PM. An adjustment of 25%, 40% and 55% is applied in the low, central and high damage cost cases respectively to the coefficient linking long-term exposure to NO_2 and mortality.

COMEAP considered that the coefficients for all-year O_3 are likely to be independent of those for either $PM_{2.5}$ or NO_2, meaning that that there is less concern about possible over-estimation when using them in a combined assessment (COMEAP, 2022b).

Table 6-1 Mapping of CRF bound chosen to each damage cost

Pollutant	Pathway	Damage cost sensitivity		
		Low	Central	High
$PM_{2.5}$	Mortality (long-term exposure)	L	C	H
$PM_{2.5}$	Respiratory hospital admission	L	C	H
$PM_{2.5}$	Cardiovascular hospital admission			C
SO_2	Mortality attributable to short-term exposure	L	C	H
SO_2	Respiratory hospital admission	L	C	H
O_3	Mortality attributable to short-term exposure	L	C	H
O_3	Respiratory hospital admission	L	C	H
O_3	Cardiovascular hospital admission			C
NO_2	Respiratory hospital admission	L	C	H
NO_2	Cardiovascular hospital admission			C
NO_2	Mortality (long-term exposure)	L	C	H
PM_{10}	Chronic Bronchitis incidence			C
$PM_{2.5}$	IHD incidence	L	C	H
NO_2	Asthma (Adults) incidence			C
$PM_{2.5}$	Stroke incidence	L	C	H
$PM_{2.5}$	Diabetes incidence			C
NO_2	Diabetes incidence			C
$PM_{2.5}$	Lung Cancer incidence	L	C	H
NO_2	Lung Cancer incidence			C
$PM_{2.5}$	Asthma (Older Children) incidence	L	C	H
NO_2	Asthma (Small Children) incidence	L	C	H
NO_2	Asthma (Older Children) incidence	L	C	H
All	Productivity	L	C	H
All	Ecosystems	L	C	H

Note: L = Low end of CRF bound; C = central point of CRF bound; H = high end of CRF bound

6.2.2 Value a proportion of deaths attributable to short-term exposure using the ‘good health VOLY’

No range is recommended by the IGCB around the value of deaths attributable to short-term exposure[15] brought forward from short term exposure and hence this value does not vary between low and high sensitivities. However, there is uncertainty around the quality of the life lost through the short-term mortality impacts of air pollutants.

As discussed in (Defra, 2007), it might be expected that deaths attributable with short-term exposure from respiratory disease occur in persons that are already ill. However, evidence suggests that for cardiovascular disease, some deaths occur in apparently healthy people (i.e. with no symptoms of prior underlying illness).

To address this uncertainty, the original damage cost report proposes that between 10 and 15% of deaths attributed to short-term exposure could therefore be valued using the ‘good health VOLY’ (value of life year lost in good health) used to value the effects of mortality associated with long-term exposure as a sensitivity. This project has included 15% of deaths attributed to short-term exposure being valued using this higher valuation in the high damage cost estimate.

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[15] Previously described as deaths brought forward

6.2.3 Life-years-lost per death attributed to short-term exposure

In order to convert the life years lost attributable as a consequence of short-term exposure to air pollution it is necessary to make an assumption around the number of months or years of life lost by an affected individual. COMEAP’s estimate of between 2 and 6 months per death is recommended by the IGCB as the best estimate to use. It is important to note that there is still uncertainty around the amount of life lost through short-term exposure and this range was mainly inferred by COMEAP from the underlying evidence base rather than being based on direct evidence (for comparison the EU CAFE approach to the estimation of impacts assumes one life-year lost per death attributed to short-term exposure).

For this project, we have followed published IGCB guidance and have assumed the lower (2 months) and higher (6 months) levels of life lost under the low and high damage cost estimate respectively. For the central estimate, the project has assumed a central value of 4 months of life lost per death.

7 Updated damage costs

7.1 2025 results

The updated set of damage costs are presented in the following tables, for the central values and the low and high sensitivities. These values represent the damage costs associated with air pollutant emissions in 2025, presented in 2025 prices. All sustained impacts of air pollutant emissions have been discounted back to the year in which the impact being assessed takes place (e.g. for a change in emissions in 2025, impacts in 2026, 2027, 2028, etc., are discounted to present day, i.e., 2025). A positive damage cost value means that an increase in pollutant emissions generates a cost to society, and a decrease in emissions generates a benefit to society (or cost decrease).

Please note that, for these damage costs, changes in PM emissions are captured in terms of $PM_{2.5}$ emissions. An adjustment is made to the PM_{10} pathways included for the ratio of primary $PM_{2.5}$ to PM_{10} emissions such that the change in emissions is expressed correctly when combined with these pathways. Ratios have been calculated from the NAEI emissions for 2022. These are presented in Section 2.5.

Note also that these damage costs have been produced applying an adjusted coefficient for long-term mortality effects associated with exposure to NO_2 following COMEAP's advice for assessing 'interventions primarily target NOx' reflecting IGCB's steer. It is important to note that strictly COMEAP's recommendation regarding the estimation of mortality effects and the overlap with PM focused only road traffic emissions. This reflects that the epidemiological evidence for the CRF comes from studies where the main driver for the spatial variation in air pollutant concentrations was emissions from road traffic). The mix of 'all pollutants' emitted for other sectors is likely to be different because for most sectors the source emitting are not engines. Thus, using the adjusted NOx coefficient applied here may be considered less applicable, increasing uncertainty of applying these damage costs.

Table 7-1 below contains the national damage costs. The damage costs for VOC include impacts via the O_3 pathways only. Table 7-2 and Table 7-3 provide sector-specific damage costs for $PM_{2.5}$ and NO_x. Section 7.4 provides a more granular breakdown of damage cost values for the sensitivities selected.

Table 7-1 Revised national damage cost estimates and sensitivity bounds (2025 prices, impacts discounted to 2025). $PM_{2.5}$ is the preferred metric for the change in PM emissions

Pollutant Emitted	Central Damage Cost (£/tonne)	Low – High damage cost sensitivity range (£/tonne)	
		Low sensitivity damage cost	High sensitivity damage cost
NOx	10,193	1,671	34,546
SO_2	26,193	9,158	72,022
NH_3	9,900	3,376	27,868
VOC	150	90	277
$PM_{2.5}$	114,411	40,026	371,654

Table 7-2 Revised sector PM damage cost estimates and sensitivity bounds (2025 prices, impacts discounted to 2025). PM2.5 is the preferred metric for the change in PM emissions

Pollutant Emitted	Central Damage Cost (£/tonne)	Low – High damage cost sensitivity range (£/tonne)	
		Low sensitivity damage cost	High sensitivity damage cost
PM2.5 Part A Category 1	12,917	4,478	36,781
PM2.5 Part A Category 2	59,137	20,486	166,364
PM2.5 Part A Category 3	230,713	80,004	659,233
PM2.5 Part A Category 4	5,329	1,848	15,289
PM2.5 Part A Category 5	8,098	2,810	23,389

Pollutant Emitted	Central Damage Cost (£/tonne)	Low – High damage cost sensitivity range (£/tonne)	
		Low sensitivity damage cost	High sensitivity damage cost
PM2.5 Part A Category 6	25,534	8,857	73,279
PM2.5 Part A Category 7	1,481	513	4,234
PM2.5 Part A Category 8	4,177	1,450	12,082
PM2.5 Part A Category 9	9,173	3,195	28,040
PM2.5 Industry (area)	170,576	61,511	787,110
PM2.5 Commercial	146,671	50,747	404,555
PM2.5 Domestic	109,172	37,747	297,946
PM2.5 Solvents	182,830	63,652	554,483
PM2.5 Road Transport	121,222	42,305	380,527
PM2.5 Aircraft	146,743	51,084	444,486
PM2.5 Offroad	89,066	30,825	246,788
PM2.5 Rail	111,755	38,690	311,294
PM2.5 Ships	65,693	22,723	180,426
PM2.5 Waste	106,862	36,976	295,121
PM2.5 Agriculture	35,416	12,934	184,157
PM2.5 Other	149,541	51,762	415,338
PM2.5 Road Transport Central London	648,110	226,292	2,048,400
PM2.5 Road Transport Inner London	628,530	219,583	2,002,690
PM2.5 Road Transport Outer London	353,225	123,383	1,123,061
PM2.5 Road Transport Inner Conurbation	253,612	88,530	798,916
PM2.5 Road Transport Outer Conurbation	159,099	55,524	499,532
PM2.5 Road Transport Urban Big	152,869	53,356	480,719
PM2.5 Road Transport Urban Large	122,884	42,888	386,116
PM2.5 Road Transport Urban Medium	99,639	34,775	313,066
PM2.5 Road Transport Urban Small	84,998	29,664	266,921
PM2.5 Road Transport Rural	49,192	17,163	153,912
PM2.5 Rail Transport Central London	558,441	193,113	1,527,561
PM2.5 Rail Transport Inner London	709,957	245,507	1,941,808
PM2.5 Rail Transport Outer London	456,691	157,929	1,249,411
PM2.5 Rail Transport Inner Conurbation	266,139	92,033	728,001
PM2.5 Rail Transport Outer Conurbation	145,097	50,181	397,628
PM2.5 Rail Transport Urban Big	158,795	54,916	434,845
PM2.5 Rail Transport Urban Large	119,759	41,449	332,076
PM2.5 Rail Transport Urban Medium	89,462	30,949	246,241
PM2.5 Rail Transport Urban Small	73,313	25,389	205,259
PM2.5 Rail Transport Rural	45,868	15,892	129,411
PM2.5 Domestic Central London	862,547	298,193	2,348,976

Pollutant Emitted	Central Damage Cost (£/tonne)	Low – High damage cost sensitivity range (£/tonne)	
		Low sensitivity damage cost	High sensitivity damage cost
PM2.5 Domestic Inner London	733,939	253,754	2,001,566
PM2.5 Domestic Outer London	394,288	136,326	1,075,708
PM2.5 Domestic Inner Conurbation	292,459	101,120	798,172
PM2.5 Domestic Outer Conurbation	182,873	63,230	499,094
PM2.5 Domestic Urban Big	156,677	54,173	427,606
PM2.5 Domestic Urban Large	143,176	49,504	390,775
PM2.5 Domestic Urban Medium	107,392	37,132	293,097
PM2.5 Domestic Urban Small	86,151	29,788	235,129
PM2.5 Domestic Rural	50,279	17,384	137,214

Table 7-3 Revised sector NOx national damage cost estimates and sensitivity bounds (2025 prices, impacts discounted to 2025).

Pollutant Emitted	Central Damage Cost (£/tonne)	Low – High damage cost sensitivity range (£/tonne)	
		Low sensitivity damage cost	High sensitivity damage cost
NOx Part A Category 1	3,612	1,035	10,972
NOx Part A Category 2	4,896	1,159	15,572
NOx Part A Category 3	10,258	1,678	34,779
NOx Part A Category 4	3,097	985	9,125
NOx Part A Category 5	3,532	1,027	10,685
NOx Part A Category 6	5,255	1,194	16,855
NOx Part A Category 7	2,961	972	8,639
NOx Part A Category 8	3,364	1,011	10,083
NOx Part A Category 9	3,650	1,038	11,105
NOx Industry (area)	12,228	1,868	41,837
NOx Commercial	21,315	2,747	74,389
NOx Domestic	18,998	2,523	66,089
NOx Solvents	21,796	2,794	76,114
NOx Road Transport	13,631	2,004	46,863
NOx Aircraft	17,172	2,346	59,547
NOx Offroad	11,431	1,791	38,982
NOx Rail	13,241	1,966	45,465
NOx Ships	3,623	1,036	11,011
NOx Waste	11,422	1,790	38,949
NOx Agriculture	4,724	1,142	14,956
NOx Other	4,442	1,115	13,945
NOx Road Transport Central London	72,737	7,721	258,606
NOx Road Transport Inner London	71,618	7,613	254,598
NOx Road Transport Outer London	40,048	4,559	141,501
NOx Road Transport Inner Conurbation	28,927	3,483	101,658
NOx Road Transport Outer Conurbation	18,365	2,462	63,823
NOx Road Transport Urban Big	17,655	2,393	61,279

Pollutant Emitted	**Central Damage Cost (£/tonne)**	**Low – High damage cost sensitivity range (£/tonne)**	
		Low sensitivity damage cost	High sensitivity damage cost
NOx Road Transport Urban Large	14,307	2,069	49,286
NOx Road Transport Urban Medium	11,630	1,810	39,695
NOx Road Transport Urban Small	10,012	1,654	33,899
NOx Road Transport Rural	5,944	1,260	19,323
NOx Rail Transport Central London	83,509	8,764	297,196
NOx Rail Transport Inner London	80,591	8,483	286,738
NOx Rail Transport Outer London	51,611	5,681	182,916
NOx Rail Transport Inner Conurbation	31,508	3,737	110,898
NOx Rail Transport Outer Conurbation	17,233	2,357	59,752
NOx Rail Transport Urban Big	18,074	2,439	62,763
NOx Rail Transport Urban Large	14,422	2,087	49,677
NOx Rail Transport Urban Medium	10,937	1,750	37,191
NOx Rail Transport Urban Small	8,867	1,551	29,774
NOx Rail Transport Rural	5,960	1,271	19,355
NOx Domestic Central London	92,180	9,612	328,231
NOx Domestic Inner London	87,005	9,113	309,690
NOx Domestic Outer London	53,260	5,849	188,798
NOx Domestic Inner Conurbation	34,167	4,003	120,397
NOx Domestic Outer Conurbation	22,755	2,900	79,511
NOx Domestic Urban Big	20,114	2,645	70,045
NOx Domestic Urban Large	17,188	2,363	59,561
NOx Domestic Urban Medium	13,097	1,968	44,902
NOx Domestic Urban Small	10,618	1,730	36,020
NOx Domestic Rural	6,403	1,323	20,916

Table 7-4 to Table 7-6 below disaggregate a selection of the damage costs by contributing impact pathways, for the central, low and high estimates separately. The tables also show that the magnitude of impacts remains broadly consistent with the 2023 update. In more detail:

- The impacts of long-term exposure to pollutants on mortality continue to be the most dominant impact valued across all damage costs.
- For the NOx damage cost, long-term exposure to PM on mortality is still an important effect (but in this case PM is a ‘secondary’ pollutant), but the mortality effect of long-term exposure to NO2 is the largest single pathway. This is the case even though the adjustment to account for the overlap between the two long-term effects has been applied to the NO2 impacts, rather than the PM effects.
- Other key impact pathways (for all damage costs) are productivity, IHD, stroke and asthma in children.
- Most other pathways are relatively small.

The balance of impacts is similar under the low damage costs. Under the high sensitivity, mortality effects associated with long-term exposure and asthma in children are important, but chronic bronchitis and diabetes pathways added under this sensitivity are also key contributors.

Where damage costs are deployed to assess impacts in years after 2025:

- No annual uplift should be applied to account for income growth between years (previously the damage and activity costs applied a 2% uplift in real terms between years, but this was changed in the 2023 update with the adoption of the following discount rate)

- Impacts in years after the first year of the appraisal period should be discounted using the Green Book discount rate for human health effects: i.e. 1.5% for year 0 to 30, 1.29% for year 31 to 75 and 1.07% for year 75 to 125 (HMT, 2024).

Table 7-4 Updated national damage costs for 2025 and contributing pathways (£2025 prices, impacts discounted to 2025) – Central

Pollutant Emitted	NOx	SO_2	NH_3	VOC	$PM_{2.5}$
Damage Cost (£/tonne)	10,193	26,193	9,900	150	114,411
$PM_{2.5}$ Mortality (long-term exposure)	1,647	15,216	5,525	0	66,886
$PM_{2.5}$ Respiratory hospital admission	8	77	28	0	338
$PM_{2.5}$ Cardiovascular hospital admission	0	0	0	0	0
PM_{10} Respiratory hospital admission	0	0	0	0	0
PM_{10} Cardiovascular hospital admission	0	0	0	0	0
SO_2 Mortality (short-term exposure)	0	67	0	0	0
SO_2 Respiratory hospital admission	0	121	0	0	0
O_3 Mortality (short-term exposure)	-12	0	0	5	0
O_3 Respiratory hospital admission	-59	0	0	22	0
O_3 Cardiovascular hospital admission	0	0	0	0	0
NO_2 Respiratory hospital admission	100	0	0	0	0
NO_2 Cardiovascular hospital admission	0	0	0	0	0
NO_2 Mortality (short-term exposure)	0	0	0	0	0
NO_2 Mortality (long-term exposure)	3,836	0	0	0	0
$PM_{2.5}$ Productivity	90	834	303	0	3,667
PM_{10} Productivity	0	0	0	0	0
O_3 Productivity	-66	0	0	25	0
O_3 Productivity	0	0	0	0	0
NO_2 Productivity	0	0	0	0	0
O_3 Material damage	-28	0	0	4	0
PM_{10} Building soiling	0	0	0	0	1,485
SO_2 Material damage	0	316	0	0	0
SO_2 Ecosystems	0	0	0	0	0
O_3 Ecosystems	-38	0	0	20	0
O_3 Ecosystems	50	0	0	75	0
NO_2 Ecosystems	142	0	0	0	0
NH_3 Ecosystems	0	0	572	0	0
PM_{10} Chronic Bronchitis Incidence	0	0	0	0	0
$PM_{2.5}$ IHD Incidence	155	1,430	519	0	6,288
NO_2 Asthma (Adults) Incidence	0	0	0	0	0
$PM_{2.5}$ Stroke Incidence	269	2,485	902	0	10,925
$PM_{2.5}$ Diabetes Incidence	0	0	0	0	0
NO_2 Diabetes Incidence	0	0	0	0	0
$PM_{2.5}$ Lung Cancer Incidence	21	190	69	0	833
NO_2 Lung Cancer Incidence	0	0	0	0	0
$PM_{2.5}$ Asthma (Children) Incidence	591	5,457	1,982	0	23,989
NO_2 Asthma (Small Children) Incidence	2,569	0	0	0	0
NO_2 Asthma (Older Children) Incidence	918	0	0	0	0

Notes: Resp. HA = Respiratory Hospital Admission; CV HA = Cardiovascular Hospital Admission

Table 7-5 Updated national damage costs for 2025 and contributing pathways (£2025 prices, impacts discounted to 2025) – Low

Pollutant Emitted	NOx	SO$_2$	NH$_3$	VOC	PM$_{2.5}$
Damage Cost (£/tonne)	1,671	9,158	3,376	90	40,026
PM$_{2.5}$ Mortality (long-term exposure)	788	7,279	2,643	0	31,996
PM$_{2.5}$ Respiratory hospital admission	-2	-17	-6	0	-75
PM$_{2.5}$ Cardiovascular hospital admission	0	0	0	0	0
PM$_{10}$ Respiratory hospital admission	0	0	0	0	0
PM$_{10}$ Cardiovascular hospital admission	0	0	0	0	0
SO$_2$ Mortality (short-term exposure)	0	34	0	0	0
SO$_2$ Respiratory hospital admission	0	41	0	0	0
O$_3$ Mortality (short-term exposure)	-2	0	0	1	0
O$_3$ Respiratory hospital admission	-8	0	0	3	0
O$_3$ Cardiovascular hospital admission	0	0	0	0	0
NO$_2$ Respiratory hospital admission	20	0	0	0	0
NO$_2$ Cardiovascular hospital admission	0	0	0	0	0
NO$_2$ Mortality (short-term exposure)	0	0	0	0	0
NO$_2$ Mortality (long-term exposure)	532	0	0	0	0
PM$_{2.5}$ Productivity	30	282	102	0	1,239
PM$_{10}$ Productivity	0	0	0	0	0
O$_3$ Productivity	-20	0	0	8	0
O$_3$ Productivity	0	0	0	0	0
NO$_2$ Productivity	0	0	0	0	0
O$_3$ Material damage	-28	0	0	4	0
PM$_{10}$ Building soiling	0	0	0	0	1,485
SO$_2$ Material damage	0	316	0	0	0
SO$_2$ Ecosystems	0	0	0	0	0
O$_3$ Ecosystems	-25	0	0	13	0
O$_3$ Ecosystems	42	0	0	62	0
NO$_2$ Ecosystems	46	0	0	0	0
NH$_3$ Ecosystems	0	0	192	0	0
PM$_{10}$ Chronic Bronchitis Incidence	0	0	0	0	0
PM$_{2.5}$ IHD Incidence	-11	-102	-37	0	-449
NO$_2$ Asthma (Adults) Incidence	0	0	0	0	0
PM$_{2.5}$ Stroke Incidence	-12	-113	-41	0	-497
PM$_{2.5}$ Diabetes Incidence	0	0	0	0	0
NO$_2$ Diabetes Incidence	0	0	0	0	0
PM$_{2.5}$ Lung Cancer Incidence	5	42	15	0	185
NO$_2$ Lung Cancer Incidence	0	0	0	0	0
PM$_{2.5}$ Asthma (Children) Incidence	151	1,397	507	0	6,142
NO$_2$ Asthma (Small Children) Incidence	167	0	0	0	0
NO$_2$ Asthma (Older Children) Incidence	0	0	0	0	0

Notes: Resp. HA = Respiratory Hospital Admission; CV HA = Cardiovascular Hospital Admission

Table 7-6 Updated national damage costs for 2025 and contributing pathways (£2025 prices, impacts discounted to 2025) – High

Pollutant Emitted	NOx	SO_2	NH_3	VOC	$PM_{2.5}$
Damage Cost (£/tonne)	34,546	72,022	27,868	277	371,654
$PM_{2.5}$ Mortality (long-term exposure)	2,286	21,124	7,670	0	92,857
$PM_{2.5}$ Respiratory hospital admission	37	344	125	0	1,511
$PM_{2.5}$ Cardiovascular hospital admission	8	71	26	0	310
PM_{10} Respiratory hospital admission	0	0	0	0	0
PM_{10} Cardiovascular hospital admission	0	0	0	0	0
SO_2 Mortality (short-term exposure)	0	115	0	0	0
SO_2 Respiratory hospital admission	0	201	0	0	0
O_3 Mortality (short-term exposure)	-35	0	0	13	0
O_3 Respiratory hospital admission	-158	0	0	59	0
O_3 Cardiovascular hospital admission	-9	0	0	3	0
NO_2 Respiratory hospital admission	240	0	0	0	0
NO_2 Cardiovascular hospital admission	113	0	0	0	0
NO_2 Mortality (short-term exposure)	0	0	0	0	0
NO_2 Mortality (long-term exposure)	10,472	0	0	0	0
$PM_{2.5}$ Productivity	200	1,849	671	0	8,126
PM_{10} Productivity	12	91	36	0	721
O_3 Productivity	-212	0	0	80	0
O_3 Productivity	-4	0	0	2	0
NO_2 Productivity	0	0	0	0	0
O_3 Material damage	-28	0	0	4	0
PM_{10} Building soiling	0	0	0	0	1,485
SO_2 Material damage	0	316	0	0	0
SO_2 Ecosystems	0	0	0	0	0
O_3 Ecosystems	-53	0	0	28	0
O_3 Ecosystems	59	0	0	88	0
NO_2 Ecosystems	328	0	0	0	0
NH_3 Ecosystems	0	0	1,412	0	0
PM_{10} Chronic Bronchitis Incidence	2,035	15,737	6,246	0	125,207
$PM_{2.5}$ IHD Incidence	472	4,359	1,583	0	19,162
NO_2 Asthma (Adults) Incidence	0	0	0	0	0
$PM_{2.5}$ Stroke Incidence	815	7,531	2,735	0	33,107
$PM_{2.5}$ Diabetes Incidence	810	7,483	2,717	0	32,892
NO_2 Diabetes Incidence	8,203	0	0	0	0
$PM_{2.5}$ Lung Cancer Incidence	43	393	143	0	1,728
NO_2 Lung Cancer Incidence	123	0	0	0	0
$PM_{2.5}$ Asthma (Children) Incidence	1,343	12,409	4,506	0	54,548
NO_2 Asthma (Small Children) Incidence	5,035	0	0	0	0
NO_2 Asthma (Older Children) Incidence	2,411	0	0	0	0

Notes: Resp. HA = Respiratory Hospital Admission; CV HA = Cardiovascular Hospital Admission

7.2 Comparison of the updated damage costs to previous estimates

For comparison, the updated central damage costs are presented alongside the original set of costs and those published in 2023, 2020, 2019 and 2015 in Table 7-7, in the ‘original’ prices, and in Table 7-8, in 2025 prices for comparability.

Table 7-7 Updated and original central damage cost estimates

Pollutant	Original damage cost (£2005/t)	Damage costs 2015 (£2015/t)	Damage costs 2019 (£2017/t)	Damage costs 2020 (£2017/t)	Damage costs 2023 (£2022/t)	Damage costs 2025 (£2025/t)
NOx National	875	*	6,199	6,385	8,148	10,193
NOx Domestic	*	14,646	13,950	12,448	12,881	18,998
NOx Industry***	*	13,131	*	*	*	*
NOx Industry (area sources) ***	*	*	5,671	5,891	8,635	12,228
NOx Road Transport	*	25,252	10,699	9,066	11,682	13,631
SO_2	1,496	1,956	6,273	13,026	16,616	26,193
NH_3	1,884	2,363	6,046	7,923	9,667	9,900
VOC	*	*	102	102	172	150
$PM_{2.5}$	*	*	105,836	73,403	74,769	114,411
$PM_{2.5}$ Domestic	25,770	33,713	85,753	89,456	84,629	109,172
$PM_{2.5}$ Industry ***	23,103	30,225	*	*	*	*
$PM_{2.5}$ Industry (area sources) ***	*	*	95,847	71,455	76,354	170,576
$PM_{2.5}$ Road Transport	44,430	58,125	203,331	81,518	84,548	121,222
$PM_{2.5}$ Waste	19,105	24,994	162,082	74,029	72,008	106,862

Table 7-8 Updated and original central damage cost estimates uplifted to 2025 prices

Pollutant	Original damage cost (£2025/t)	Damage costs 2015 (£2025/t)	Damage costs 2019 (£2025/t)	Damage costs 2020 (£2025/t)	Damage costs 2023 (£2025/t)	Damage costs 2025 (£2025/t)
NOx National	1,462	*	8,199	8,445	9,348	10,193
NOx Domestic	*	20,118	18,450	16,464	14,777	18,998
NOx Industry***	*	18,037	*	*	*	*
NOx Industry (area sources) ***	*	*	7,501	7,792	9,906	12,228
NOx Road Transport	*	34,687	14,151	11,991	13,402	13,631
SO_2	2,499	2,687	8,297	17,228	19,062	26,193
NH_3	3,148	3,246	7,997	10,479	11,090	9,900
VOC	*	*	135	135	197	150
$PM_{2.5}$	*	*	139,980	97,084	85,777	114,411
$PM_{2.5}$ Domestic	43,054	46,309	113,418	118,316	97,088	109,172
$PM_{2.5}$ Industry ***	38,598	41,518	*	*	*	*
$PM_{2.5}$ Industry (area sources) ***	*	*	126,768	94,507	87,595	170,576
$PM_{2.5}$ Road Transport	74,230	79,842	268,928	107,817	96,996	121,222
$PM_{2.5}$ Waste	31,919	34,332	214,372	97,912	82,609	106,862

* = no damage cost estimated
** NOx damage costs presented are those 'where PM not valued'
*** Between the 2015 and updated damage costs there was a slight adjustment to the coverage of the 'industry' damage cost. The 2015 costs aggregated point and area sources, whereas the updated damage cost only focuses on area sources as point sources are separated out in the 'Part A' damage costs.

As shown above, the updated damage costs show variance from both the original and latest published sets of damage costs.

The relatively high level of inflation between 2022 and 2025 means that the 2025 damage costs would be higher than the 2023 damage costs by about 15% in the absence of any other changes. A small increase in UK population by about 2% and a larger increase in baseline death rate of about 7% also lead to a systematic increase in the contribution of the most important mortality and morbidity pathways to the overall damage costs.

There were no changes in the pathways included in the damage costs or in the methods used for each pathway apart from the updating of baseline rates, population and prices.

The air dispersion and chemistry transport models that underpin the relationships between emissions and concentrations have also been updated for the 2025 damage costs. The changes from the 2023 damage cost, therefore, vary by pollutant and sector.

For NOx:

- The 2025 damage costs have increased by 9% compared to the 2023 damage cost (expressed in 2025 prices).
- This small increase is the result of a small increase in the μgm^{-3} changes in NO_2 concentrations per tonne of NOx emission and a smaller decrease in the μgm^{-3} changes in secondary $PM_{2.5}$ concentrations per tonne of NOx emission. These small changes offset resulting in little change to the national damage cost for NOx. The change largely reflects the changes in death rate and population.

For SO_2:

- The 2025 damage costs have increased by 37% compared to the 2023 damage cost (expressed in 2025 prices).
- This increase is due to an increase in the μgm^{-3} of secondary $PM_{2.5}$ per tonne of SO_2 emitted. This is because while UK SO_2 emissions declined between 2020 and 2023, there was a small increase in measured SO_4 particulate matter over this period. The increases in population and death rates will also have contributed to this increase.

For NH_3:

- The damage cost is now 11% lower than the 2023 update (expressed in 2025 prices).
- This change is due to a reduction in the μgm^{-3} of secondary $PM_{2.5}$ per tonne of NH_3 emitted. This is because while UK NH_3 emissions were largely unchanged between 2020 and 2023, there was a decrease in measured $NH4_4$ particulate matter over this period. The reduction is partly offset by increases in population and death rates.

For VOC:

- The value in the 2025 damage costs has decreased by 24%.
- The decrease is caused by the change in relationships between VOC emissions and O_3 concentrations. These relationships have been recalculated for the 2025 damage costs using the chemistry transport model CMAQ. These changes are likely to reflect a combination of changes in atmospheric composition between the previous modelling year of 2014 the new modelling year of 2022 and differences in model formulations and assumptions.

For $PM_{2.5}$:

- The $PM_{2.5}$ damage cost increased by 33% relative to the central 2023 set (expressed in 2025 prices).
- This increase is the result of an increase in the μgm^{-3} change in primary $PM_{2.5}$ concentrations per tonne of $PM_{2.5}$ emission. This increase is primarily associated with a reduction in the estimates of UK $PM_{2.5}$ emissions between the 2019 NAEI that was used for the 2023 damage costs and the 2022 NAEI that was used for the 2025 damage costs. Revisions to the NAEI reflect changes in assumptions regarding activity levels, emission factors and the mapping of emissions.
- The $PM_{2.5}$ damage cost for industry area sources increased by 95% relative to the 2023 damage cost value for this sector. This was largely because of changes in emission factor assumptions for solid and liquid fuels for the ‘other industrial combustion’ source within the emission inventory.

7.3 Costs per μgm^{-3} concentration change

Damage costs are typically expressed per tonne of emission. However, costs can also be expressed for many of the damage cost pathways per μgm^{-3}, which is the unit of concentration. Costs expressed in this way can be used in policy appraisal. Generating these ‘direct impact’ costs essentially involves removing the first step of the method that is used to calculate the damage costs, which is the relationship between 1 tonne of emission and the concentration impact. Instead, costs are calculated for a 1 μgm^{-3} change in concentration.

The ability to calculate costs based on concentrations depends on the availability of specific detailed concentration modelling. However, where such modelling is available, deploying costs per μgm^{-3} concentration change can facilitate a more detailed and robust assessment (more akin to deploying the full IPA, where typically the most important difference relative to deploying a damage cost approach will be undertaking detailed dispersion modelling to produce a more relevant and robust picture of exposure in the appraisal domain).

Not all impacts which make up the damage costs are calculated based on concentration exposure. Some are carried through from underlying studies and estimations, and as such are deployed on a per tonne basis and not re-estimated in detail as part of developing the damage costs. As such, where the costs per change in concentration are deployed to estimate effects, analysts will also need to calculate the costs for additional ‘other impacts’ that are expressed per tonne. These ‘other impacts’ costs include pathways to assess environmental, productivity, and building impacts, and are separate to, and thus do not overlap with those included as ‘direct impacts’.

Emissions of NO_x, SO_2 and NH_3 contribute to the damage costs through the $PM_{2.5}$ concentration pathways due to the formation of secondary $PM_{2.5}$ from these emissions. If the air pollution modelling that has been used to calculate the impact of changes in emissions on $PM_{2.5}$ concentrations includes the changes in secondary $PM_{2.5}$, then these contributions will feed through into the total costs via these calculated $PM_{2.5}$ μgm^{-3} concentration changes. If, however, only the impacts of changes in primary $PM_{2.5}$ emissions have been taken into account in the modelling of $PM_{2.5}$ concentrations, then the costs related to the ‘Secondary $PM_{2.5}$ impacts’ of the changes in NO_x, SO_2 and NH_3 emissions will need to be accounted for separately based on the change in tonnes emitted.

The ‘direct impact’ (column A below) and ‘other impact’ (column C below) cost estimates can be added together to generate the total cost estimate if the formation of secondary $PM_{2.5}$ has been included in the air pollution modelling (A+C).

If the formation of secondary $PM_{2.5}$ has not been included in the air pollution modelling, then the ‘direct impact’ (column A below), ‘secondary $PM_{2.5}$’ (column B below) and ‘other impact’ (column C below) cost estimates should be added together to generate the total cost estimate (A+B+C).

The formation of secondary $PM_{2.5}$ is not typically included in the air pollution dispersion modelling that is carried out within an appraisal of local air quality policies. Inclusion of the formation of secondary pollutants such as $PM_{2.5}$ generally requires the use of a chemistry transport model in addition to an air dispersion model and is more likely to be included within the appraisal for national or regional air quality policies. The formation of O_3 is not normally included in the modelling required for an air quality appraisal, only makes a relatively modest contribution to the total damage costs and requires the calculation of complex metrics. Costs per μgm^{-3} are therefore not provided for O_3 and are included in the per tonne ‘other impacts’

This update presents a set of costs per μgm^{-3} concentration change. These are consistent with the damage costs per tonne. The values of the ‘direct impacts’ per μgm^{-3}, the ‘secondary $PM_{2.5}$’ and ‘other impacts’ that are expressed per tonne of emission are presented in Table 7-9 below. Note that the direct impacts are per μgm^{-3} of NO_2 in the first row of the table.

Table 7-9 Costs per µgm^{-3} concentration change – Central damage cost sensitivity

Primary Pollutant	Sector	A) Direct impacts - (£2025 per popwm 1 ugm^{-3} change per person per year)	B) Secondary $PM_{2.5}$ impacts (£2025 / tonne)	C) Other impacts (£2025 / tonne)
NO_X*	National	8.58	2,780	-12
SO_2	National	0.16	25,689	316
NH_3	National	-	9,328	572
VOC	National	-	-	150
$PM_{2.5}$	National	65.95	-	-

* values are expressed per µgm^{-3} of NO_2 concentration for A) and per tonne of NO_x emitted for B) and C)

Notes on using the costs per µgm^{-3} change in concentration

The costs per µgm^{-3} change in concentration are developed in a specific way and, hence, should be applied by following specific steps. In particular:

- Costs are expressed per 1 µgm^{-3} change in population-weighted concentrations of NO_2, SO_2 and $PM_{2.5}$ – as such these should be applied once population weighting has been applied to any concentration modelling.

- They are expressed per 1 µgm^{-3} change per person – i.e. these costs have been calculated per 1 µgm^{-3} change in a given pollutant on a national scale, then divided by the UK population. This has been done such that the impacts can easily be scaled to the relevant appraisal domain. As such the damage costs per change in concentration need to be multiplied by the population-weighted change in concentration, and the total population in the appraisal domain (i.e. the sum of population in the air quality modelling domain, over which the population-weighted concentrations have been calculated) to estimate the total cost.

- If the formation of secondary $PM_{2.5}$ has been included in the air pollution modelling, then the 'direct impact' and 'other impact' damage cost estimates are added together to generate the total cost estimate. (A+C). If the formation of secondary $PM_{2.5}$ has not been included in the air pollution modelling, then the 'direct impact', 'secondary $PM_{2.5}$' and 'other impact' cost estimates can be added together to generate the total cost estimate (A+B+C). Both options provide a full assessments with no double counting.

- Care needs to be taken to check whether changes in precursor emissions NO_x, SO_2 and NH_3 leading to the formation of secondary $PM_{2.5}$ are included in the per 1 µgm^{-3} $PM_{2.5}$ concentrations or need to be included per tonne of precursor emissions (secondary $PM_{2.5}$ impacts (B), above).

- The direct impacts of NO_x are calculated per 1 µgm^{-3} change in population-weighted concentration of NO_2. The secondary and other impacts are calculated per tonne of NO_x emitted.

7.4 Sensitivity analysis of damage cost values including granular breakdown

Sensitivity analysis has been conducted to consider the effect of known uncertainties on the damage cost estimates and present a more robust understanding of the evidence. The use of sensitivity boundaries is important in the application of damage costs, given the methodologies for assessing the different pathways are based on a number of assumptions.

The latest publication of damage costs includes total values for the low, central and high sensitivities. The low sensitivity incorporates both the lower bound CRF values and the lower bound for unit values for the valuation of health impacts, and a similar approach is used for the central and high values. This means that the low and high boundaries currently reflect the lowest or highest possible damage cost values, given the available evidence, and thus have very low probability of occurrence limiting their usefulness. This approach is captured in from Table 7-1.

In the 2025 damage cost update, additional sensitivity analysis has been conducted by testing the impact of adjusting assumptions across three key parameters on damage cost values. These parameters are 1) CRF values (lower, central and upper bounds), 2) unit value for impact valuation (lower, central and upper bounds), and 3) LYL per death attributed to air pollution exposure. Such an approach allows us to test the uncertainties and assumptions that have relatively larger impacts on damage cost estimates.

Table 7-10, Table 7-11 and Table 7-12 present the variation of national damage cost values by adjusting two sensitivity parameters at a time. For each table, one of the three sensitivity parameters is held at its central value, while combinations of the other two parameters are explored. This allows a more in-depth exploration of the sensitivity of the damage cost values to parameter values and uncertainties. In fact, the results of this exercise show that damage cost estimates are most sensitive to CRF assumptions, followed by the unit values for impact valuation, and least sensitive to ‘LYL per attributable death’ assumptions.

To allow comparison and avoid overestimating the effect of the CRF assumption, tables testing the ‘high CRF’ assumption include an extra row showing damage costs without the impact pathways that are considered only in the high-sensitivity scenario. (For these pathways that are only included in the high sensitivity the central DRF estimate is used.) Even with these additional pathways excluded, damage cost estimates remain most sensitive to CRF assumptions.

Table 7-10 Revised national damage cost estimates and sensitivity bounds (2025 prices, impacts discounted to 2025). Sensitivity combinations of CRF and Unit Value for Impact Valuation, with Central LYL as given. PM2.5 is the preferred metric for the change in PM emissions

		Pollutant Emitted	Unit Value for Impact Valuation		
			Low Unit Value	Central Unit Value	High Unit Value
CRF	CRF - Lower Bound	NOx	1,669	2,850	3,765
		SO_2	9,192	14,718	18,424
		NH_3	3,376	5,733	7,890
		VOC	91	117	144
		$PM_{2.5}$	40,026	63,968	79,905
	CRF - Central Estimate	NOx	5,784	10,193	13,264
		SO_2	15,667	26,193	33,262
		NH_3	5,727	9,900	13,278
		VOC	116	150	187
		$PM_{2.5}$	68,492	114,411	145,132
	CRF - Higher Bound	NOx	14,500	26,443	34,560
		SO_2	31,110	55,574	71,974
		NH_3	11,537	20,970	27,868
		VOC	181	225	272
		$PM_{2.5}$	157,708	285,908	371,654
	CRF - Higher Bound (Additional impact pathways considered only in the high sensitivity scenario are excluded)	NOx	10,287	17,997	23,277
		SO_2	22,387	38,116	48,684
		NH_3	8,170	14,232	18,880
		VOC	180	223	269
		$PM_{2.5}$	98,355	167,146	213,245

Table 7-11 Revised national damage cost estimates and sensitivity bounds (2025 prices, impacts discounted to 2025). Sensitivity combinations of CRF and LYL, with Central Unit Value for Impact Valuation as given. $PM_{2.5}$ is the preferred metric for the change in PM emissions

		Pollutant Emitted	LYL per death attributed to short-term exposure		
			Low LYL per death attributed to short-term exposure	Central LYL per death attributed to short-term exposure	High LYL per death attributed to short-term exposure
CRF	CRF - Lower Bound	NOx	2,853	2,850	2,848
		SO_2	14,685	14,718	14,752
		NH_3	5,733	5,733	5,733
		VOC	116	117	117
		$PM_{2.5}$	63,968	63,968	63,968
	CRF - Central Estimate	NOx	10,199	10,193	10,187
		SO_2	26,160	26,193	26,227
		NH_3	9,900	9,900	9,900
		VOC	148	150	153
		$PM_{2.5}$	114,411	114,411	114,411
	CRF - Higher Bound	NOx	26,453	26,443	26,433
		SO_2	55,541	55,574	55,608
		NH_3	20,970	20,970	20,970
		VOC	222	225	229
		$PM_{2.5}$	285,908	285,908	285,908
	CRF - Higher Bound (Additional impact pathways considered only in the high sensitivity scenario are excluded)	NOx	18,007	17,997	17,986
		SO_2	38,083	38,116	38,150
		NH_3	14,232	14,232	14,232
		VOC	220	223	227
		$PM_{2.5}$	167,146	167,146	167,146

Table 7-12 Revised national damage cost estimates and sensitivity bounds (2025 prices, impacts discounted to 2025). Sensitivity combinations of Unit Value for Impact Valuation and LYL, with Central CRF as given. PM2.5 is the preferred metric for the change in PM emissions

Unit Value for Impact Valuation			LYL per death attributed to short-term exposure		
		Pollutant Emitted	Low LYL per death attributed to short-term exposure	Central LYL per death attributed to short-term exposure	High LYL per death attributed to short-term exposure
	Low Unit Value	NOx	5,791	5,784	5,778
		SO_2	15,634	15,667	15,701
		NH_3	5,727	5,727	5,727
		VOC	114	116	118
		$PM_{2.5}$	68,492	68,492	68,492
	Central Unit Value	NOx	10,199	10,193	10,187
		SO_2	26,160	26,193	26,227
		NH_3	9,900	9,900	9,900
		VOC	148	150	153
		$PM_{2.5}$	114,411	114,411	114,411
	High Unit Value	NOx	13,271	13,264	13,258
		SO_2	33,229	33,262	33,296
		NH_3	13,278	13,278	13,278
		VOC	184	187	189
		$PM_{2.5}$	145,132	145,132	145,132

7.5 Intermediate health metrics

The publication of the 2023 damage costs (and preceding damage cost publications) comprised total national damage costs and total damage costs for specific sectors (and release characteristics for industrial point sources and geographical splits for road transport and rail transport sources).

In the 2025 update, the IGCB recommended the publication of intermediate health metrics, capturing the underlying health effects of air pollution in non-monetary terms, such as LYL lost, QALY lost and numbers of hospital admissions per tonne of emissions of air pollution. These metrics represent the intermediate health outcomes determined as part of (or underpinning) the damage cost calculations. More precisely, the intermediate metrics are:

- Mortality – LYL lost per tonne of emissions
- Morbidity – QALY lost per tonne of emissions
- Morbidity – Number of hospital admissions per tonne of emissions

These intermediate metrics are presented for each of the health pathways as national values (including low and high sensitivities).

The calculation of these intermediate metrics is based upon the damage cost values presented previously, which embed an analysis of health impacts over the lifetime of individuals exposed to air pollution, to determine the impact associated with exposure in a given year. To derive estimates for the underlying metrics from the damage cost values, the damage cost values (£/tonne) have been divided either the monetary value of a LYL (£/LYL) or a hospital admission (£/admission) as applicable. This calculation results in estimates of LYL/tonne, QALY lost/tonne or admissions/tonne of emissions. All discounting (as applied to the damage cost values) has been removed as the health metrics do not represent an economic valuation.

These intermediate metrics have been calculated for each of the PM2.5, NO2 and O3 health pathways. The contributions to $PM_{2.5}$, NO_2 and O_3 health pathways as a result of emissions of SO2, NH3 and VOC have been included within the estimations. The outputs of these calculations are presented in the following tables. More specifically, Table 7-13, Table 7-14, and Table 7-15 provide the intermediate metrics for each pathway aggregated by output metric for the central, low and high sensitivity criteria. The health pathways which include multiple pollutants reflect the aggregated impact of the individual pollutant pathways.

Table 7-16, Table 7-17, and Table 7-18 provide a granular breakdown per pathway.

Please also note that some of the damage cost values are negative due to negative CRF lower bound values and/or increases in NOx emissions leading to reductions in O_3 concentration. This also affects the values for the intermediate metrics.

Table 7-13 Undiscounted intermediate health metrics – Central values

Health Pathway	Metric	Units	Pollutant Emitted				
			NOx	SO_2	NH_3	VOC	$PM_{2.5}$
PM2.5 + NO2	Mortality attributable to long-term exposure	LYL/tonne	0.113	0.313	0.114	-	1.376
PM2.5 + NO2	Morbidity attributable to long-term exposure	QALY lost/tonne	0.068	0.136	0.049	-	0.596
PM2.5 + NO2 + O3	Hospital admissions	Admissions/tonne	0.004	0.007	0.002	0.002	0.030
O3	Mortality attributable to short-term exposure	LYL/tonne	- 0.0004	-	-	0.0002	-

Table 7-14 Undiscounted intermediate health metrics – Low values

Health Pathway	Metric	Units	Pollutant Emitted				
			NOx	SO_2	NH_3	VOC	$PM_{2.5}$
PM2.5 + NO2	Mortality attributable to long-term exposure	LYL/tonne	0.043	0.237	0.086	-	1.043
PM2.5 + NO2	Morbidity attributable to long-term exposure	QALY lost/tonne	0.009	0.038	0.014	-	0.166
PM2.5 + NO2 + O3	Hospital admissions	Admissions/tonne	0.003	-0.005	-0.002	0.001	-0.020
O3	Mortality attributable to short-term exposure	LYL/tonne	-0.0001	-	-	0.00003	-

Table 7-15 Undiscounted intermediate health metrics – High values

Health Pathway	Metric	Units	Pollutant Emitted				
			NOx	SO_2	NH_3	VOC	$PM_{2.5}$
PM2.5 + NO2	Mortality attributable to long-term exposure	LYL/tonne	0.211	0.349	0.127	-	1.533
PM2.5 + NO2	Morbidity attributable to long-term exposure	QALY lost/tonne	0.217	0.470	0.175	-	2.562
PM2.5 + NO2 + O3	Hospital admissions	Admissions/tonne	0.012	0.022	0.008	0.003	0.098
O3	Mortality attributable to short-term exposure	LYL/tonne	-0.001	-	-	0.0004	-

Table 7-16 Undiscounted intermediate health metrics for 2025 and contributing pathways – Central values

Pollutant emitted	Units	NOx	SO_2	NH_3	VOC	$PM_{2.5}$
PM2.5 Mortality attributable to long-term exposure	LYL/tonne	0.034	0.313	0.114	-	1.376
PM2.5 Respiratory hospital admission	Admissions/tonne	0.001	0.007	0.002	-	0.030
PM2.5 Cardiovascular hospital admission	Admissions/tonne					
O3 Mortality attributable to short-term exposure	LYL/tonne	-0.0004	-	-	0.0002	-
O3 Respiratory hospital admission	Admissions/tonne	-0.005	-	-	0.002	-
O3 Cardiovascular hospital admission	Admissions/tonne					
NO2 Respiratory hospital admission	Admissions/tonne	0.009	-	-	-	-
NO2 Cardiovascular hospital admission	Admissions/tonne					
NO2 mortality attributable to long-term exposure	LYL/tonne	0.079	-	-	-	-
PM10 Chronic Bronchitis	QALY lost/tonne	-	-	-	-	-
PM2.5 IHD	QALY lost/tonne	0.002	0.018	0.007	-	0.079
NO2 Asthma (Adults)	QALY lost/tonne	-	-	-	-	-
PM2.5 Stroke	QALY lost/tonne	0.004	0.033	0.012	-	0.143
PM2.5 Diabetes	QALY lost/tonne	-	-	-	-	-
NO2 Diabetes	QALY lost/tonne	-	-	-	-	-
PM2.5 Lung Cancer	QALY lost/tonne	0.0002	0.002	0.001	-	0.010
NO2 Lung Cancer	QALY lost/tonne	-	-	-	-	-
PM2.5 Asthma (Children)	QALY lost/tonne	0.009	0.083	0.030	-	0.364
NO2 Asthma (Small Children)	QALY lost/tonne	0.039	-	-	-	-
NO2 Asthma (Older Children)	QALY lost/tonne	0.014	-	-	-	-

Table 7-17 Undiscounted intermediate health metrics for 2025 and contributing pathways – Low values

Pollutant emitted	Units	NOx	SO_2	NH_3	VOC	$PM_{2.5}$
PM2.5 mortality attributable to long-term exposure	LYL/tonne	0.026	0.237	0.086	-	1.043
PM2.5 Respiratory hospital admission	Admissions/tonne	-0.0005	-0.005	-0.002	-	-0.020
PM2.5 Cardiovascular hospital admission	Admissions/tonne	-	-	-	-	-
O3 Mortality attributable to short-term exposure	LYL/tonne	-0.0001	-	-	0.00003	-
O3 Respiratory hospital admission	Admissions/tonne	-0.002	-	-	0.001	-
O3 Cardiovascular hospital admission	Admissions/tonne	-	-	-	-	-
NO2 Respiratory hospital admission	Admissions/tonne	0.005	-	-	-	-
NO2 Cardiovascular hospital admission	Admissions/tonne	-	-	-	-	-
NO2 mortality attributable to long-term exposure	LYL/tonne	0.017	-	-	-	-
PM10 Chronic Bronchitis	QALY lost/tonne	-	-	-	-	-
PM2.5 IHD	QALY lost/tonne	-0.0003	-0.003	- 0.001	-	- 0.011
NO2 Asthma (Adults)	QALY lost/tonne	-	-	-	-	-
PM2.5 Stroke	QALY lost/tonne	-0.0003	-0.003	- 0.001	-	- 0.013
PM2.5 Diabetes	QALY lost/tonne	-	-	-	-	-
NO2 Diabetes	QALY lost/tonne	-	-	-	-	-
PM2.5 Lung Cancer	QALY lost/tonne	0.0001	0.001	0.000	-	0.004
NO2 Lung Cancer	QALY lost/tonne	-	-	-	-	-
PM2.5 Asthma (Children)	QALY lost/tonne	0.005	0.042	0.015	-	0.186
NO2 Asthma (Small Children)	QALY lost/tonne	0.005	-	-	-	-
NO2 Asthma (Older Children)	QALY lost/tonne	-	-	-	-	-

Table 7-18 Undiscounted intermediate health metrics for 2025 and contributing pathways – High values

Pollutant emitted	Units	NOx	SO_2	NH_3	VOC	$PM_{2.5}$
PM2.5 mortality attributable to long-term exposure	LYL/tonne	0.038	0.349	0.127	-	1.533
PM2.5 Respiratory hospital admission	Admissions/tonne	0.002	0.018	0.007	-	0.081
PM2.5 Cardiovascular hospital admission	Admissions/tonne	0.0004	0.004	0.001	-	0.016
O3 Mortality attributable to short-term exposure	LYL/tonne	-0.001	-	-	0.0004	-
O3 Respiratory hospital admission	Admissions/tonne	-0.008	-	-	0.003	-
O3 Cardiovascular hospital admission	Admissions/tonne	-0.0005	-	-	0.0002	-
NO2 Respiratory hospital admission	Admissions/tonne	0.013	-	-	-	-
NO2 Cardiovascular hospital admission	Admissions/tonne	0.006	-	-	-	-
NO2 mortality attributable to long-term exposure	LYL/tonne	0.173	-	-	-	-
PM10 Chronic Bronchitis	QALY lost/tonne	0.018	0.140	0.055	-	1.111
PM2.5 IHD	QALY lost/tonne	0.004	0.041	0.015	-	0.181
NO2 Asthma (Adults)	QALY lost/tonne	-	-	-	-	-
PM2.5 Stroke	QALY lost/tonne	0.008	0.074	0.027	-	0.325
PM2.5 Diabetes	QALY lost/tonne	0.008	0.070	0.026	-	0.310
NO2 Diabetes	QALY lost/tonne	0.077	-	-	-	-
PM2.5 Lung Cancer	QALY lost/tonne	0.0004	0.004	0.001	-	0.015
NO2 Lung Cancer	QALY lost/tonne	0.001	-	-	-	-
PM2.5 Asthma (Children)	QALY lost/tonne	0.015	0.141	0.051	-	0.621
NO2 Asthma (Small Children)	QALY lost/tonne	0.057	-	-	-	-
NO2 Asthma (Older Children)	QALY lost/tonne	0.027	-	-	-	-

Table 7-19 below presents the percentage of total damage costs captured by the sum of the PM, NO_2 and O_3 health pathways. This table shows that these health pathways capture more than 90% and the majority of the total damage costs. However, there are other pathways that also contribute to damage costs, and non-health pathways, which have not been included in this intermediate metric analysis. Values in this table were calculated using discounted values to ensure consistency with the damage costs presented in £.

Table 7-19 Percentage total 2025 £ per tonne of emissions national damage costs represented by the sum of the PM, NO_2 and O_3 health pathways

Pollutant sensitivity ranges	PM, NO_2 and O_3 Health pathways percentage of total damage cost
NOx national_Low Sensitivity	97%
PM2.5 national_Low Sensitivity	93%
NOx national_Central Sensitivity	99%
PM2.5 national_Central Sensitivity	95%
NOx national_High Sensitivity	99%
PM2.5 national_High Sensitivity	97%

Finally, potential users of these metrics should note that their application is a less precise and less scientifically robust method to establish the potential health effects, in non-monetary terms, of air pollution, especially when compared to the impact pathway approach (IPA). The latter directly relies on the assessment of human exposure to pollutant concentrations in air and the direct application of concentration response functions (CRFs) under local circumstances. These intermediate ‘underlying health’ metrics should be used for the purposes of a simplified assessment to approximate the health impacts of marginal changes in air pollution in the absence of concentration data, and considered to provide a sense of direction and magnitude of impact. That is, these metrics can provide a suitable approximation and provide an evidence-based narrative to accompany the valuation of air pollution impacts using damage costs.

8 Updated activity costs

The updated activity costs are presented in the tables below. There are no domestic emissions for some fuels in some areas within the NAEI 2022 mapped emissions inventory. It is not possible to calculate an activity cost for these fuels in these areas. This is indicated as “N/A” in Table 8-2.

Where activity costs are deployed to assess impacts in years after 2025:

- No annual uplift should be applied to account for income growth between years (previously the damage and activity costs applied a 2% uplift in real terms between years, but this has changed with the adoption of the following discount rate)

- Impacts in years after the year of emissions change should be discounted using the Green Book discount rate for human health effects: i.e. 1.5% for year 0 to 30, 1.29% for year 31 to 75 and 1.07% for year 75 to 125 (HMT, 2024).

Table 8-1 Transport activity costs (p/litre; impacts in 2025 in £2025 prices; update to BEIS’s ‘Table 14’)

Area	Car petrol	Car diesel	LGV petrol	LGV diesel	Rigid HGV diesel	Articulated HGV diesel
Transport average	1.97	12.37	2.22	13.73	7.52	2.16
Central London	10.42	65.88	11.78	73.14	40.02	11.37
Inner London	10.25	64.78	11.59	71.94	39.35	11.18
Outer London	5.74	36.25	6.49	40.26	22.03	6.27
Inner conurbation	4.15	26.18	4.69	29.07	15.91	4.53
Outer conurbation	2.64	16.62	2.98	18.46	10.10	2.89
Urban big	2.54	15.97	2.87	17.74	9.71	2.78
Urban large	2.06	12.94	2.33	14.38	7.87	2.25
Urban medium	1.68	10.52	1.89	11.69	6.40	1.84
Urban small	1.45	9.06	1.63	10.06	5.51	1.58
Transport rural	0.86	5.38	0.98	5.98	3.28	0.95

Table 8-2 National and domestic activity costs (p/kWh; impacts in £2025 prices; update to BEIS's 'Table 15')

Sector	Fuel	2025	2030	2035	2040	2045	2050
NATIONAL AVERAGE	Electricity	0.187	0.083	0.022	0.006	0.001	0.001
	Gas	0.195	0.195	0.195	0.195	0.195	0.195
	Coal	4.898	4.898	4.898	4.898	4.898	4.898
	Oil	2.179	2.179	2.179	2.179	2.179	2.179
	Biomass	2.862	2.862	2.862	2.862	2.862	2.862
	LPG	0.417	0.417	0.417	0.417	0.417	0.417
	Peat	34.061	34.061	34.061	34.061	34.061	34.061
	Petroleum coke	9.841	9.841	9.841	9.841	9.841	9.841
DOMESTIC: Inner conurbation	Gas	0.279	0.279	0.279	0.279	0.279	0.279
	Coal	23.272	23.272	23.272	23.272	23.272	23.272
	Oil	0.982	0.982	0.982	0.982	0.982	0.982
	Biomass	44.107	44.107	44.107	44.107	44.107	44.107
	LPG	N/A	N/A	N/A	N/A	N/A	N/A
	Peat	N/A	N/A	N/A	N/A	N/A	N/A
	Petroleum coke	47.614	47.614	47.614	47.614	47.614	47.614
DOMESTIC: Urban big	Gas	0.171	0.171	0.171	0.171	0.171	0.171
	Coal	15.403	15.403	15.403	15.403	15.403	15.403
	Oil	0.434	0.434	0.434	0.434	0.434	0.434
	Biomass	23.724	23.724	23.724	23.724	23.724	23.724
	LPG	N/A	N/A	N/A	N/A	N/A	N/A
	Peat	46.917	46.917	46.917	46.917	46.917	46.917
	Petroleum coke	43.828	43.828	43.828	43.828	43.828	43.828
DOMESTIC: Urban medium	Gas	0.112	0.112	0.112	0.112	0.112	0.112
	Coal	12.672	12.672	12.672	12.672	12.672	12.672
	Oil	0.286	0.286	0.286	0.286	0.286	0.286
	Biomass	16.274	16.274	16.274	16.274	16.274	16.274
	LPG	N/A	N/A	N/A	N/A	N/A	N/A
	Peat	32.477	32.477	32.477	32.477	32.477	32.477
	Petroleum coke	42.480	42.480	42.480	42.480	42.480	42.480
DOMESTIC: Urban small	Gas	0.092	0.092	0.092	0.092	0.092	0.092
	Coal	11.414	11.414	11.414	11.414	11.414	11.414
	Oil	0.228	0.228	0.228	0.228	0.228	0.228
	Biomass	13.078	13.078	13.078	13.078	13.078	13.078
	LPG	N/A	N/A	N/A	N/A	N/A	N/A
	Peat	N/A	N/A	N/A	N/A	N/A	N/A
	Petroleum coke	41.897	41.897	41.897	41.897	41.897	41.897
DOMESTIC: Rural	Gas	0.069	0.069	0.069	0.069	0.069	0.069
	Coal	9.407	9.407	9.407	9.407	9.407	9.407
	Oil	0.162	0.162	0.162	0.162	0.162	0.162
	Biomass	7.655	7.655	7.655	7.655	7.655	7.655
	LPG	0.054	0.054	0.054	0.054	0.054	0.054
	Peat	14.882	14.882	14.882	14.882	14.882	14.882
	Petroleum coke	40.892	40.892	40.892	40.892	40.892	40.892

Appendix 1 – References

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