Air Quality damage cost update 2023 – FINAL Report

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 economic and/or social costs (known as external costs or externalities) that are not captured in the market price of the goods or services consumed that produce the pollution. The UK Department of Environment, Food and Rural Affairs (Defra) has produced guidance (Defra, 2021b) 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 details two approaches to assessing and valuing the impacts: one is the 'damage cost' approach, which is to be taken where impacts are valued to be less than £50m and when compliance with legally binding objectives is expected. Damage costs are a set of impact values defined per tonne of emission. These values estimate the external costs associated with a marginal change in pollutant emissions. They can be combined with forecasts of emission changes to provide an approximate valuation of the aggregate external impacts of a policy.

Ricardo Energy & Environment has been commissioned by Defra to update the damage costs of air pollution. A number of tasks were included in this update, including:

1. Update the emissions-to-concentration modelling to 2020.
2. Update several CRFs (including that linking mortality to chronic exposure to $PM_{2.5}$) to reflect latest COMEAP guidance.
3. Aligning with the updated HMT Green Book, in particular with relation to discounting and value of a QALY.
4. Adding new rail damage costs split by area type.
5. Update to all underlying population and baseline health data.

The updated set of damage costs is presented in Table 1-1 below, alongside the low and high estimated sensitivities around the central values. This table shows the national average damage costs. Sector specific damage costs have also been updated and are also presented in this report. A positive damage cost represents a cost associated with an increase in pollutant emissions or a benefit associated with a decrease in pollutants emissions.

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

Each of the updated damage costs shows some variation relative to the set of damage costs published in 2020 (Ricardo, 2020). 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. Specific changes in the individual damage costs, and key drivers are as follows:

- The damage costs for NOx have increased in comparison to the damage costs published in 2020. A key driver is the increase in the CRF for mortality associated with long-term exposure to $PM_{2.5}$ and an upward revision of the ugm^{-3} change in secondary inorganic aerosol (SIA) per tonne of NO_X, resulting in an increase in QALY loss assigned.

- For $PM_{2.5}$, the national average damage cost has decreased by 10% relative to the central 2020 set. A key change has been the update to the CRF for IHD incidence associated with chronic exposure to reflect the latest COMEAP advice. Likewise, the CRF for stroke incidence associated with long-term exposure has also been revised downwards. These changes have been partly (but not wholly) offset by the increase in the CRF for mortality associated with long-term exposure to $PM_{2.5}$ and update to the QALY value.

- For SO_2, the national damage cost is higher than in the 2020 damage costs. This is a result of the increase in the CRF used for mortality associated with long-term exposure to $PM_{2.5}$. Updating the QALY loss value has also marginally increased the damage costs.
- For VOC, damage costs have increased relative to the 2020 set. The increased is related to updating ecosystem valuations for ozone impacts on livestock production and ozone impacts of CO2 sequestration to reflect the latest feed prices and carbon price.
- For NH_3, the damage cost is now slightly higher than the 2020 update. The key driver here has been driven by the increase in the CRF for mortality associated with long-term exposure to $PM_{2.5}$, updating the QALY loss value and the change to select only 'robust' ecosystem pathways.

Although the damage costs have been revised to reflect specific improvements in the underlying evidence base, the guidance regarding their use is still appropriate. The damage costs should only be used in appraisal where the cumulative monetised impacts sum to less than £50m or where the impacts are ancillary. 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 the wider caveats around their use, in particular regarding the uncertainty associated with their estimation and the coverage of impacts included and 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.
- 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.

Table 1-1 -Revised national average damage cost estimates and sensitivity bounds (2022 prices, 2022 impact year). $PM_{2.5}$ is the preferred metric for the change in PM emissions

Pollutant Emitted	Central Damage Cost (£/t)	Low – High damage cost sensitivity range (£/t)	
		Low sensitivity damage cost	High sensitivity damage cost
NOx	8,148	1,567	30,282
SO_2	16,616	6,615	43,850
NH_3	9,667	3,727	26,172
VOC	172	104	309
$PM_{2.5}$	74,769	29,631	212,839

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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, 2021) 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, 2022) 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 details two approaches to assessing and valuing the impacts of policy on air quality. It recommends analysts follow the 'damage cost' approach where impacts are valued to be less than £50m and the more rigorous 'impact-pathway' approach (IPA) where impacts are more significant.

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. Hence it is recommended that the damage costs are only used for narrowing down a long list of policy options (before undertaking more detailed assessment) or for policy appraisal where either the air pollution impacts are secondary, or the total level of impacts is valued to be less than £50m.

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 chronic 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 chronic exposure to NO_2 to reflect updated COMEAP guidance
- Inclusion of new impact pathways: chronic 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 (using NAEI 2018) for: NO_x and NO_2, $PM_{2.5}$ and PM_{10} and SO_2.

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 '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'.

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. The scope of the project included the following key actions:

- Update to the emissions-to-concentrations modelling for all pollutants except ozone
- Updates to the concentration-response-function (CRF) for a selection of impact pathways, following updated guidance issued by COMEAP (most importantly updating the CRF linking chronic exposure to $PM_{2.5}$ to mortality)
- Splitting rail 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)
- Selected update to the ecosystem impacts included, to account for developments in the valuation of impacts since 2014
- Complementary update to the air pollution activity costs.

Although the scope of the impacts has been expanded as part of this update, it is important to note that the damage costs 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 et al. (2014) – those ranked as 'robust' and 'acceptable'. However, pathways assessed as 'improvements desirable' have not been included.
- The damage costs for VOC 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 presents some comparisons with previous versions
- Section 8 concludes by presenting the updated activity costs.

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 has not been updated

- Relationship between changes in NO_x emissions and ambient O_3 concentrations (Ozone Source Receptor Model (OSRM) model)
- Relationship between changes in VOC emissions and ambient O_3 concentrations (OSRM model).

We would not expect the O_3 modelling to show large changes from the modelling included in the 2020 damage costs, which was based on the NAEI 2013. Furthermore the O_3 pathways do not make a dominant contribution to the overall damage costs. As such these have not been updated for this revision of the damage costs. We would recommend that the O_3 modelling be updated once every ~10 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 2020 on a 1 x 1 km grid. This model has been described in detail by (Pugsley, et al., 2022).

Overall, the modelling approach adopted is very similar to that used for the 2020 damage costs, which were based on air quality modelling for 2018. The changes in the emissions to concentrations relationships derived from the 2018 air quality modelling largely therefore reflect changes in the input datasets for the modelling relative to 2018. The inputs for the 2022 modelling include emission inventory estimates from the 2019 NAEI and ambient air quality measurement and meteorological data for 2020.

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 2020 were calculated for each pollutant using 1 x 1km population data from the 2011 census updated to be applicable for 2019. The μgm^{-3} per tonne for each pollutant was then calculated by dividing this population-weighted mean by the 2020 UK total emissions for each pollutant that were used to calculate the ambient concentrations within the model. The emissions for 2020 were calculated by scaling data from the NAEI for 2019 forwards by one year using emission projections provided by the NAEI as described by (Pugsley, et al., 2022). Specific adjustments to the emissions inventory inputs for the 2020 dispersion modelling were required to account for the reduced activity levels during 2020 as a result of the covid-19 pandemic related lockdowns. These adjustments (described in detail by (Pugsley, et al., 2022)) enable the models to provide an unbiased assessment of concentration during 2020 and ensure that the emissions to concentrations relationships derived from this modelling remain reliable.

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 2-1. 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 (Brookes, et al., 2020)). 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 emissions sources have also been separated by area type.

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 2-1 - 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

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 2-2.

Table 2-2 - 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 2-3.

Table 2-3 – 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., 2022).

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 capturinRomeTW45

g 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 OSRM method for impact of changes in NO_x and VOC emissions on O_3

The modelling for O_3 has not been updated for the 2023 damage costs and is unchanged from the modelling used for the 2020 damage costs. The modelling for the 2020 damage costs was based on estimates of concentrations in 2014 and calculated based on emissions inventory estimates for 2013.

The Ozone Source Receptor Model (OSRM) was used to calculate the impact of changes in NO_x emissions and VOC (non-methane VOC) emissions on ambient O_3 concentrations. The modelled change in μgm^{-3} O_3 per tonne of NO_x emissions or VOC emissions was then used in the impact pathways for O_3 concentrations.

The OSRM was run to model the impact of a 10% reduction in UK NO_x emissions on O_3 concentrations on a 10 km x10 km UK grid. The impact of this scenario on various population-weighted mean ozone O_3 metrics (for the UK) was then calculated from the gridded results. The model was also run to assess the impact of a 10% reduction in UK VOC emissions on O_3 concentrations. The μgm^{-3} changes per tonne changes in emissions were then calculated by dividing the changes in the population-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. A reduction in VOC emissions results in a reduction in O_3 concentration. A reduction in NO_x emissions results in an increase in O_3 concentration.

Emissions of different VOC species have differing potentials to influence photochemical ozone creation. A consistent reduction in all VOC species has been assumed in the calculation of the impact of VOC emissions via the O_3 impact pathways. Consideration of damage costs for different VOC species is beyond the scope of this update to the damage costs.

Further information on the OSRM can be found in (Cooke et al, 2014).

2.5 Dispersion modelling to support estimation of ecosystem impacts

The modelling for O_3 has not been updated for the 2023 damage costs and is unchanged from the modelling used for the 2020 damage costs. The modelling for the 2020 damage costs was based on estimates of concentrations in 2014 and calculated based on emissions inventory estimates for 2013.

Specific O_3 concentration metrics were required to include the valuation of the three ecosystem impacts from O_3. 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}) and 24-hour mean averaged over a seven-month growing season from 1st March to 30th September.

To produce the POD_6 wheat metric, additional post-processing of OSRM results was carried out. This separate post-process was run on the OSRM model results from the impact of a 10% reduction in UK NO_x emissions on O_3 concentrations on a 10 km x 10 km UK grid. It was also run on the OSRM model results from the 10% reduction in UK VOC emissions scenario. The impact of these scenarios on area weighted mean POD_6wheat were then calculated from the results (following the method described in section 2.4).

Separate 7-month (rather than 12-month) OSRM runs were carried out to produce the 7-month 24-hour mean metric. 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 concentration were then calculated from the results (following the method described in section 2.4).

Further information on the OSRM can be found in (Cooke et al, 2014).

2.6 Specific changes to concentration modelling for this update

There are few systematic changes to the dispersion modelling for 2020 compared to the dispersion modelling for 2018 (which was used for the 2020 damage cost update). The following changes may be relevant to the damage cost emissions to concentrations modelling:

- The application of scaling factors 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.

- The use of specific 2020 traffic counts for major roads and traffic statistics derived from observations in 2020 to calculate emissions from major and minor roads respectively. Previous assessments have applied scaling factors to account for the relatively small changes in traffic activity between the emission inventory year (2019 in this instance) and the concentration year (2020). Traffic activity levels were unusually low during 2020 due to the covid-19 lockdowns. The direct use of 2020 data to model 2020 concentrations should lead to a reduction in uncertainty and have little overall impact on the emission to concentrations relationships.

- The methods used to map the emissions estimates across the UK are subject to routine updates and improvements. There were no changes in methods between the 2017 and 2019 inventories that are likely to have large systematic impact on the emission to concentrations relationships.

- The emissions estimates for road transport included in the 2020 modelling used updated assumption on vehicle speeds and a revised method for assignment of average speed based on speed limits. This change is not expected to have a large impact on the emission to concentrations relationships.

- The dispersion modelling for 2020 used meteorological data from the WRF meteorological model for 50 km squares across the UK. Previous assessments used observed meteorological data for a single location. The dispersion modelling is calibrated using observed ambient concentrations. This change is not expected to have a large impact on the emission to concentrations relationships.

Emission sensitivity coefficients have been used to calculate the impact of 10% reductions of UK NO_x, SO_2 and NH_3 emissions on population-weighted mean annual mean SIA concentrations in the UK. 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. The method has been revised for 2020 to ensure that only emissions likely to be relevant to the formation of UK SIA are included in the UK total emissions estimates. Published emissions inventories tend to include emissions in the national total, such as ‘international cruise’ for aircraft, which are assigned to a specific country but are not relevant for SIA formation in the UK. These emissions have been excluded in the calculation of the 2020 damage costs. This results in an increase in the calculated concentration change per tonne emitted, particularly for NOx.

2.7 PM conversion factors

Note that for these damage costs the change in $PM_{2.5}$ emission is the preferred metric for PM emissions. The IGCB 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 IGCB 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.

Hence 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 2019 for each area type – these are presented in the table below. The value for UK total emissions ($PM_{2.5}/PM_{10}$) is 0.646. Sector specific ratios have been used for the individual emissions sectors and these vary from 0.176 for agriculture to 0.995 for off-road mobile machinery.

Table 2-4 – $PM_{2.5}$-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	0.646	Road Transport Central London	0.647
Part A Category emissions		Road Transport Inner London	0.647
Part A Category 1	0.822	Road Transport Outer London	0.623
Part A Category 2	0.807	Road Transport Inner Conurbation	0.626
Part A Category 3	0.785	Road Transport Outer Conurbation	0.622
Part A Category 4	0.803	Road Transport Urban Big	0.625
Part A Category 5	0.709	Road Transport Urban Large	0.628
Part A Category 6	0.762	Road Transport Urban Medium	0.630
Part A Category 7	0.795	Road Transport Urban Small	0.631
Part A Category 8	0.783	Road Transport Rural	0.616
Part A Category 9	0.596	Rail Transport Central London	0.953
Area source sector emissions		Rail Transport Inner London	0.953
Industry (area)	0.383	Rail Transport Outer London	0.952
Commercial	0.961	Rail Transport Inner Conurbation	0.953
Domestic	0.978	Rail Transport Outer Conurbation	0.944
Solvents	0.651	Rail Transport Urban Big	0.949
Road Transport	0.622	Rail Transport Urban Large	0.910
Aircraft	0.819	Rail Transport Urban Medium	0.933
Offroad	0.995	Rail Transport Urban Small	0.871
Rail	0.894	Rail Transport Rural	0.866
Ships	0.947		
Waste	0.932		
Agriculture	0.176		
Other	0.897		

3 Estimation and valuation of human health impacts

3.1 Introduction

For the 2023 damage cost update, changes were made to the methods and assumptions used to estimate the human health impacts. The updated approach and the unchanged methods relative to the 2020 damage cost update are 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 damage costs calculations have been updated to reflect a number of different developments in the underlying evidence base. A key update has been to reflect recent publications by COMEAP around the quantification of impacts. Specifically, the following concentration response functions have been updated reflecting updates to COMEAP guidance:

- Update of the estimation of mortality effects of chronic exposure to $PM_{2.5}$ (COMEAP, 2022).
- Update of the estimate of respiratory and cardiovascular hospital admission effects associated with short-term exposure to PM (COMEAP, 2022b). Key changes are that CRFs are now expressed for $PM_{2.5}$ rather than PM_{10}. Cardiovascular hospital admission pathways are now removed to avoid overlaps with morbidity effects associated with chronic exposure included.
- Update of the estimate of hospital admission effects of exposure to NO_2 (COMEAP, 2022b) (although noting this pathway is only included in the 'high' sensitivity damage cost). Evidence for respiratory effects of NO_2 following short-term exposure has increased.
- Update of the estimate of incidence of cardiovascular disease and stroke associated with long term exposure to $PM_{2.5}$ (COMEAP, 2021). Updated recommendation is for Ischaemic Heart Disease (IHD) rather than Coronary Heart Disease (CHD), but noting that the difference in definition of IHD and CHD is not significant for the estimation of effects in this case.

The following wider updates were also made to the workbook to update the estimation of human health impacts.

- Update population data to reflect the latest ONS population estimates (ONS, 2020b).
- Update England, Wales and Northern Ireland baseline mortality rates and hospital admission rates with the latest information (ONS, 2020; NISRA, 2020; National Records of Scotland, 2020)
- Update UK asthma rates with the latest information (British Lung Foundation, 2018; Asthma.org, 2018), and an adjustment to the data source used to define baseline incidence of IHD (BHF, 2012).
- Update QALY to align with the latest guidance in the Green Book 2022 (HM Treasury, 2022; Arden, 2013).

3.3 Concentration response functions (CRFs) for health outcomes

3.3.1 CRFs carried forward from previous COMEAP guidance

The estimation of the impacts of air pollution is carried out using Concentration Response Functions (CRFs). CRFs link a change in exposure to a pollutant to its consequent impacts by expressing a change in a health (or non-health) outcome for a given change in pollutant concentrations.

In its 2013 published guidance, the Interdepartmental Group on Costs and Benefits (IGCB) has recommended a set of CRFs describing the health impacts of air pollution that it suggests should be used for the appraisal of air quality impacts (Defra, 2013). These CRFs were taken from an extensive underlying literature 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 are carried forward to the updated damage costs.

COMEAP have subsequently published a number of 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 deaths brought forward (COMEAP, 2015b).
- Statement on quantifying mortality associated with long-term average concentrations of fine particulate matter ($PM_{2.5}$) (COMEAP, 2018b)

The CRFs for these pathways carried forward used for the estimation of the updated damage costs are set out in Table 3-1.

Table 3-1 – CRFs applied in updated damage costs (% per 10µgm^{-3} change in concentration for relevant averaging period)

			% change per 10ugm^{-3} change in pollutant		
Pollutant	Pathway	Air pollution metric	Low	Central	High
SO_2	Deaths brought forward (1)	Annual average	0.6	0.6	0.6
SO_2	Respiratory hospital admission (1)	Annual average	0.5	0.5	0.5
O_3	Deaths brought forward (2)	Daily maximum of 8 hour mean	0.12	0.34	0.56
O_3	Respiratory hospital admission (2)	Daily maximum of 8 hour mean	0.3	0.75	1.2
O_3	Cardiovascular hospital admission (2)	Daily maximum of 8 hour mean	-0.06	0.11	0.27
NO_2	Mortality associated with long-term exposure (3)	Annual average	0.8 (0.2)	2.3 (0.92)	3.7 (2.035)
PM_{10}	Chronic bronchitis (4)	Annual average	1.02	1.32	1.71

* Pathway only for inclusion in sensitivity analysis. CRFs with the adjustment for overlap with $PM_{2.5}$ applied are included in brackets. Source: (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.

In its guidance, IGCB did not include a sensitivity range around the CRFs linking short-term, acute exposure to SO_2 to hospital admissions or mortality. Hence the CRFs used to assess these impacts are not flexed to derive the 'low' and 'high' damage cost sensitivities.

In the previous estimation of damage costs, 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 project, based on the most recent advice from COMEAP regarding the estimation of effects associated with ozone exposure (COMEAP, 2015) we have not applied a threshold to the calculation of effects across all damage cost sensitivities.

3.3.2 CRFs carried forward from PHE (morbidity associated with chronic 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 provided by Public Health England (PHE), which in turn were obtained from scientific papers.

The health outcomes included and associated CRFs are:

- *Asthma in adults.* CRF for NO_2, with a value of Odds Ratio (OR) of 1.04 (0.996; 1.08) per 10 µg/m^3 of annual mean. Data sourced from (Jacquemin, et al., 2015).
- *Asthma in small children (≤ 6 years old)*. CRF for NO_2, with a value of Odds Ratio (OR) of 1.08 (1.01, 1.12) per 10 µg/m^3 of annual mean. Data sourced from (Khreis, et al., 2016).
- *Asthma in older children (7-15 years old).* CRF for NO_2, with a value of Odds Ratio (OR) of 1.03 (1.00, 1.06) per 10 µg/m^3 of annual mean; CRF for $PM_{2.5}$, with a value of Odds Ratio (OR) of 1.48 (1.22, 1.97) per 10 µg/m^3 of annual mean. Data sourced from (Khreis, et al., 2016).
- *Diabetes Type 2.* CRF for $PM_{2.5}$, with a value of Relative Risk (RR) of 1.10 (1.02; 1.18) per 10 µg/m^3 of annual mean. CRF for NO_2, with a value of Relative Risk (RR) of 1.05 (1.02; 1.07) per 10 µg/m^3 of annual mean. Data sourced from (Eze, et al., 2015).
- *Lung cancer.* CRF for $PM_{2.5}$, with a value of Relative Risk of 1.09 (1.04; 1.14) per 10 µg/m^3 of annual mean. CRF for NO_2 with a value of Relative Risk (RR) of 1.02 (1.00; 1.03) per 10 µg/m^3 of annual mean. Data sourced from (Hamra, et al., 2015)

CRFs were also defined for CHD and stroke, but these have been subsequently updated in this 2023 update (see Section 3.3.3.3), and hence the previous CRF values are omitted from the list above.

All CRFs are assumed to represent a change in incidence, as suggested by most of the references that were used in the PHE report. 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 applied these CRFs in a micro-simulation model to estimate the implied changes in disease incidence. The updated damage costs are not based on this type of micro-simulation model, and calculated values for the impact of change in concentrations on incidence from the PHE model have not been made available. Instead, we have assumed that the CRFs for all the diseases can be assumed to apply directly to incidence for the purpose of calculating damage costs (also for simplification, these calculations do not include a (cessation) lag to represent the delay in effects associated with chronic exposure). The CRFs are summarised in Table 3-2. We have included only the pathways that are considered more certain in the central damage costs as shown in Table 3-6 below.

Table 3-2 – CRF's applied in updated damage costs (% per defined change in concentration for relevant averaging period) – PHE morbidity pathways

					% change (or change in Odds Ratio) per defined change in pollutant		
Pollutant	Pathway	Air pollution metric	CRF type	Reference change in concentration (µgm^{-3})	Low	Central	High
$PM_{2.5}$	Diabetes	Annual average	Relative Risk (RR)	10	2.00	10.00	18.00
NO_2	Diabetes	Annual average	Relative Risk (RR)	10	2.00	5.00	7.00
$PM_{2.5}$	Lung cancer	Annual average	Relative Risk (RR)	10	4.00	9.00	14.00
NO_2	Lung cancer	Annual average	Relative Risk (RR)	10	0.00	2.00	3.00
NO_2	Asthma (Adults)	Annual average	Odds Ratio (OR)	10	1.00	1.04	1.08
$PM_{2.5}$	Asthma (Older Children)	Annual average	Odds Ratio (OR)	10	1.22	1.48	1.97
NO_2	Asthma (Small Children)	Annual average	Odds Ratio (OR)	10	1.01	1.08	1.12
NO_2	Asthma (Older Children)	Annual average	Odds Ratio (OR)	10	1.00	1.03	1.06

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 in turn have been collected 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[1].
- *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).

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 1 was applied:

$$I_i = \sum_j^J \sum_k^K \frac{N_{j,k}}{N} \cdot I_{i,j,k} \quad (1)$$

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 2:

$$\Delta I_i = \frac{\Delta C_{Pol}}{C_{Inc}} \cdot RR \cdot \frac{N}{10^5} \cdot I_i \quad (2)$$

Where:

[1] 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

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:

$$\kappa_i = \exp\left(-\ln(OR)\cdot\frac{\Delta C_{Pol}}{C_{Inc}} + \ln\frac{I_i}{10^5 - I_i}\right) \qquad (3)$$

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 4:

$$\Delta I_i = \frac{N(1+\kappa_i)}{\kappa_i(I_i - 1) + I_i} \qquad (4)$$

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 Updates to CRFs for 2023 damage costs

3.3.3.1 Mortality effects of chronic exposure to $PM_{2.5}$

Since the 2020 guidance, COMEAP have issued updated guidance around the assessment of mortality effects associated with long-term exposure to $PM_{2.5}$. This took the form of the publication 'Statement on quantifying mortality associated with long-term exposure to $PM_{2.5}$' (COMEAP, 2022). Recommendations for quantification of mortality associated with long-term average concentrations of exposure to NO_2 and O_3 remain as before.

COMEAP's updated guidance is based on the summary effects from a meta-analysis of the available global literature. The guidance recommends a CRF of 1.08 (95% CI: 1.06, 1.09) per 10 µg/m^3 annual average $PM_{2.5}$ and assumes continuing linearity[2] when quantification is performed down to very low or even zero $PM_{2.5}$ concentrations. It should be noted that the recommended CRF is not adjusted for effects of other pollutants[3], which means that:

- mortality estimates will likely include effects caused by other correlated pollutants (such as NO_2) to some extent and

[2] 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.

[3] There are a number of challenges in interpreting the results of two-pollutant models. COMEAP (2018a; section 3.2.3) summarises the statistical issues as including: the lack of an interaction term; multi-collinearity (high correlations between pollutant concentrations); transfer of effect arising from exposure misclassification; and overlapping confidence intervals between coefficients reported from single- and two-pollutant models. In addition a coefficient for $PM_{2.5}$, even when adjusted for another pollutant (such as NO_2), likely reflects the effects of other pollutants which are more closely correlated with $PM_{2.5}$ than the other pollutant (NO_2 in this example) to some extent (COMEAP, 2018a table 7.1)

- if mortality effects estimated using this coefficient are added to estimates of mortality effects associated with other pollutants, this will likely give an overestimate of the effects of the pollution mixture and of the benefits of reducing concentrations.

The CRFs used for the estimation of the updated damage costs are set out in Table 3-3.

Table 3-3 - CRF's applied in updated damage costs (% per 10µgm-3 change in concentration for relevant averaging period) - mortality associated with long-term exposure to $PM_{2.5}$

			% change per 10ugm^{-3} change in pollutant		
Pollutant	Pathway	Air pollution metric	Low	Central	High
$PM_{2.5}$	Mortality associated with long-term exposure	Annual average	6	8	9

3.3.3.2 Respiratory and cardiovascular hospital admission effects of exposure to $PM_{2.5}$ and NO_2

COMEAP have also published a report exploring the link between exposure to $PM_{2.5}$ and NO_2 and hospital admissions: Statement on update of recommendations for quantifying hospital admissions associated with short-term exposures to air pollutants (COMEAP, 2022b). In the statement, COMEAP considered summary effects estimates (coefficients) from single pollutant models derived in meta-analyses of the global literature (undertaken by St George's, University of London with funding from the Department of Health) as the most suitable for use as CRFs to quantify hospital admissions associated with short-term exposures to air pollutants.

A key update for the 2023 damage costs is that hospital admissions associated with particulate exposure are associated now with $PM_{2.5}$ rather than PM_{10}.

COMEAP recommended that the 24-hour effect estimates for NO_2 are used in health impact assessments of interventions to improve air quality. Concentrations of $PM_{2.5}$ and NO_2 are often highly correlated, meaning that associations reported from epidemiological studies likely reflect the effect of both pollutants to some extent. Therefore, using coefficients for both $PM_{2.5}$ and NO_2 (for the same health end-point) within the same assessment would result in an over-estimation of the effect of the air pollution mixture, or of the benefits of interventions to reduce emissions. 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. In addition, policy-makers should be aware that localised interventions designed to reduce NO_2 may have the unintended consequence of increasing localised concentrations of O_3.

In this statement, COMEAP also drew attention to the uncertainties regarding causality for some pollutant-outcome pairs, notably cardiovascular hospital admissions associated with NO_2; these uncertainties will need to be considered when deciding which pollutant-outcome pairs to include in core assessments or in sensitivity analyses. In addition, COMEAP also drew attention to the potential for overlap between effects associated with chronic and acute exposure (e.g. hospital admissions).

Based on discussions with UKHSA as part of the present damage cost update, several methodological choices have been implemented:

- Cardio-vascular hospital admissions associated with NO_2 would appear only in the high damage cost sensitivity
- Indeed, all cardiovascular hospital admissions have been removed from low and central damage costs sensitivity scenarios due to the potential overlap with chronic cardiovascular conditions. The current approach to quantifying cardiovascular hospital admissions attributable to short-term exposures to $PM_{2.5}$ are based on coefficients from time-series studies. It is not clear to what extent these avoided hospital admissions are a subset of the cardiovascular benefits of reductions in long-term exposure to $PM_{2.5}$. The central estimate for cardiovascular hospital admissions is now only included in the high damage cost sensitivity scenario.
- There is also the potential for overlap between chronic respiratory conditions and respiratory hospital admissions. However, the risk of overlap was considered lower for respiratory relative to cardiovascular conditions, in particular given respiratory conditions associated with chronic exposure, such as chronic bronchitis and asthma in adults, are only included in the high sensitivity

damage cost. In addition, hospital admissions are still a small effect. As such respiratory hospital admissions were retained in the central (and low) damage costs.

The CRFs used for the estimation of the updated damage costs are set out in Table 3-4.

Table 3-4 - CRFs applied in updated damage costs (% per 10µgm-3 change in concentration for relevant averaging period) – respiratory hospital admissions and CV hospital admissions for $PM_{2.5}$ and NO_2

Pollutant	Pathway	Air pollution metric	% change per 10ugm^{-3} change in pollutant	
			Effect estimate	95% CI
$PM_{2.5}$	Respiratory hospital admissions, all ages	24-hour average	0.96	-0.63 to 2.58
$PM_{2.5}$	CV hospital admissions, all ages	24-hour average	0.90	0.26 to 1.53
NO_2	Respiratory hospital admissions, all ages	24-hour average	0.57	0.33 to 0.82
NO_2	CV hospital admissions, all ages	24-hour average	0.66	0.32 to 1.01

3.3.3.3 Cardiovascular disease and stroke associated with long-term exposure to $PM_{2.5}$

Based on recommendations from COMEAP (COMEAP, 2021), the CRFs for IHD and stroke set out in Table 3-5 have been adopted in the analysis. These update the previous CRFs originally taken from the PHE study.

Table 3-5 - CRFs applied in updated damage costs (% per 10µgm-3 change in concentration for relevant averaging period) – incidence of ischemic heart disease and stroke for $PM_{2.5}$

Pollutant	Pathway	Air pollution metric	% change per 10ugm^{-3} change in pollutant	
			Effect estimate	95% CI
$PM_{2.5}$	Incidence of ischemic heart disease	24-hour average	7	-1 to 16
$PM_{2.5}$	Stroke incidence	24-hour average	11	-1 to 25

3.3.4 Health impact pathway summary

As described above, the health impact pathways included in the updated damage costs come from a number of sources. These are based on the set of health impact pathways and CRFs 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 3-6. Please note: low, central and high in the table do not 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. Which impact pathways are included, and which CRF is selected from the underlying confidence interval in each damage cost is presented in Table 6-1.

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 3-6. Primary effects, such as the mortality associated with long-term exposure to $PM_{2.5}$ 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’.

As shown in the table below, 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.

Table 3-6 – CRFs applied in updated damage costs (% per 10µgm^{-3} change in concentration for relevant averaging period)

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

Table 3-7 – 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	Deaths brought forward	Respiratory hospital admission	Respiratory hospital admission	Mortality (long-term exposure)	Productivity	Deaths brought forward	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
Damage cost	SO_2	2	2		2	2*	P	P										P					2		2	2*		2		2		
Damage cost	NH_3	2	2		2	2*																P	2		2	2*		2		2		
Damage cost	VOC											2	2		2*	2				2												
Damage cost	$PM_{2.5}$	P	P		P	P*											P						P		P	P*		P		P		

P = primary effect; 2 = secondary effect; * = impact only included under 'high' damage cost

3.4 Mortality associated with chronic exposure and life-table calculations

3.4.1 Methodology for calculating long-term air pollution impacts

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 chronic exposure to $PM_{2.5}$ and NO_2.

A life-table is a technique 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[4] system. Calculations were based on mid-year population estimates and mortality rates for 2012[5]. 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 1µgm^{-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.

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

[4] For further detail, see: http://www.iom-world.org/research/research-expertise/statistical-services/iomlifet/

[5] 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.

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.

An uplift applied has been applied to lifetable outputs to adjust for changes in population and mortality rate since the original analysis was performed.

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 either 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, 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

As described above, 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 3-8 (results show cumulative life years gained across all age cohorts, sexes and calendar years from 2012 to 2112). A value for a $1\mu gm^{-3}$ 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 3-8 – Life years gained by UK 2020 population aged 30+ from a $1\mu gm^{-3}$ one-year pulse reduction in $PM_{2.5}$

Parameter	Result
Total life years gained	49,627
(Range from low to high CRF bounds)	(37,620 – 55,273)

3.5 Baseline population and health response rates

As part of the re-estimation of the damage costs, this project has updated the population and baseline health outcomes data used in the calculation to reflect the latest available data.

Population data for the UK and each of the Devolved Administrations was taken from ONS’s mid-year population estimates for 2020 (ONS, 2021b). This represents resident population, which is also used when modelling exposure. Hence the calculations were based on a UK population of around 67.1m, of which 12.5m were over the age of 65.

Data for the number of deaths in the UK for 2019 were aggregated from data for individual Devolved Administrations sourced from the (ONS, 2020), (NISRA, 2020) and (National Records of Scotland, 2020)– note 2019 data for mortality is used given the unprecedented effect of the COVID-19 pandemic in 2020. 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, 2020) for England, (DHSSPSNI, 2020) for Northern Ireland and NHS Wales (NHS Wales Informatics Service , 2020). 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 2019/20.

Baseline rates for the prevalence and incidence of morbidity effects associated with chronic exposure are gathered from a range of sources:

- *Chronic bronchitis.* In the 2020 damage costs, chronic bronchitis was expressed as cases of chronic phlegm, and baseline data on cases of chronic phlegm in never-smokers (aged 16 and above) were taken from the same sources used by (COMEAP, 2016)) in their calculation of chronic bronchitis effects in order to be consistent with their recommendations[6].
- *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[7].
- *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).

The data provided above was per 100,000 persons of each age group. Where applicable, to obtain an age- and gender-weighted incidence of a disease *i*, Equation 1 was applied:

$$I_i = \sum_j^J \sum_k^K \frac{N_{j,k}}{N} \cdot I_{i,j,k} \qquad (1)$$

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*.

[6] COMEAP in turn adopted baseline prevalence rates for chronic phlegm in England from the Health Survey for England (HSE, 2011) and in Scotland from the Scottish Health Survey (Scottish Government, 2011)
[7] https://www.ucl.ac.uk/pcph/research-groups-themes/thin-pub/database

The risk rates used in the estimation of the damage costs are presented in Table 3-9 below.

Table 3-9 – Health outcome risk rates used for damage cost estimation (number of cases per 100,000 of population per annum)

Metric	Risk rate all ages	Risk rate in ages 16+	Risk rate in ages 65+	Risk rate 0-5	Risk rate 6-15
Deaths	858	-	-	-	-
Cardio vascular hospital admission	1,104	-	4,222	-	-
Respiratory hospital admission	1,995	-	5,401	-	-
Chronic bronchitis	-	4,967	-	-	-
Asthma in adults	-	186	-	-	-
Asthma in children	-	-	-	929	461
IHD	171	-	-	-	-
Stroke	133	-	-	-	-
Diabetes (Type II)	570	-	-	-	-
Lung cancer	78.4	-	-	-	-

3.6 Impacts on productivity

(Ricardo-AEA, 2014) explored the impacts of air pollution on productivity. 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 chronic and acute 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 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 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 pathways already included in the existing damage costs are included 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 low, central and high damage costs are:

- absenteeism and work-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 also includes impacts on school days lost (SDL) (and consequent effect of absent workers to care for dependents) through exposure to PM_{10} and O_3.

Table 3-10: 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)

A low, central and high estimate of the additional productivity impacts 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 3-10.

3.7 Valuation of health outcomes

To value the impacts of exposure to air pollutants, the estimated quantity of health effects are combined with a monetary impact value of a single instance of each health impact. In this case, this is a value of life-year lost (or VOLY, used to monetise mortality impacts), a QALY value and a value per hospital admission. Many of the impact values used for this updated set of damage costs are the same as those used in the calculation of the original damage costs, although some targeted improvements have been made.

3.7.1 Mortality associated with chronic exposure (VOLY)

The impact 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 assessment 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 2022).

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 Webtag databook (DfT, 2022) 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 0.9% per annum from 2002 to 2022. Hence using the assumed uplift could have led to substantial overestimation of the value of impacts.

The annual rate of real GDP per capita growth sourced from Webtag were used to uplift WTP estimates between the 2002 and the assessment year 2022. The price base of the VOLY estimates was updated using the latest set of GDP deflators published by (HMT, 2022b).

3.7.2 Acute morbidity (Hospital admissions)

A similar approach was adopted to value the morbidity pathways as mortality associated with long-term exposure. We have started with the original unit values used in the original estimation of damage costs, and applied uplifts using the latest data on real GDP per capita growth. These health 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 chronic exposure (QALYs)

The damage costs include a number of morbidity pathways associated with chronic exposure, inspired by the approach taken by PHE. In order 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 monetary value of a QALY.

For the current update, the monetary WTP value for a QALY in the workbook has been updated to £70,000 in 2021 prices to reflect the latest Green Book 2022 guidance (HM Treasury, 2022).

The calculation of QALY loss requires utility weights for the different diseases, which are then multiplied by the change in incidence as in Equation 5:

$$QALY\ Loss_i = (1 - w_i) \cdot \delta_i \cdot \Delta I_i \qquad (5)$$

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 2020 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 3-11, alongside their source.

Table 3-11 – 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 6:

$$\delta_i = 1 \qquad \text{if } D = 1 \qquad (6a)$$

$\delta_i = 1 + \sum_{j=2}^{D}(1+r)^{1-j}$ if $D > 1$ (6b)

Where:

D is the average years of duration of the disease.

r is the discount rate (r=0.035).

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 3-12. 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 3-12 – Average and discounted duration of disease

Disease	D [years]	δ [years]
IHD (angina)	9.50	8.93
IHD (AMI)	1	1
Asthma in Adults	23.60	20.11
Asthma in Children	36.20	28.39
Stroke	14.80	13.41
Diabetes	9.10	8.58
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 7 (see Table 3-14):

$$Cost_i = QALY\ Value \cdot QALY\ Loss_i \quad (7)$$

Following discussion with PHE regarding the strength of the underpinning epidemiological evidence, these chronic morbidity pathways were included in the damage costs (and the sensitivity around central values) as set out in Table 3-13. No pathways are included in the low damage cost.

Table 3-13 – 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

Several impact pathways will have lasting effects after the first year of impact. This applies to mortality effects associated with long-term exposure, but also some of the morbidity impacts that are measured as changes in incidence. For analysis of future year impacts, the previous approach to assume a proxy for income growth is the long-run rate of economic growth of 2% per annum has been removed as the updated

Green Book discount rates already considers the 2% uplift. Impacts in years after the year of emissions change are discounted using the Green Book 2022 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, 2022).

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 any other impact pathways given impacts occur in year in which change in emissions occur. Specifically:

1. Productivity 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
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 discounted already in the recommended values.

3.7.5 Summary of unit impact values

The unit impact values used in the analysis are set out in Table 3-14. In variance to the previous derivation of the original damage cost estimates, this project 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 £7,000 and £7,100 for respiratory and cardiovascular hospital admissions respectively.

Table 3-14 – Health impact values used in analysis (£2022 prices) *

Health effect	Form of measurement valuations apply to	Health values used in analysis (Sensitivity range)	2020 update (Defra, 2021) (Sensitivity range)
Deaths brought forward (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	£26,200 (10-15% of LYL valued using 'good health' VOLY)	£24,500 (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.	£50,600 (£37,900 – £63,100)	£47,600 (£35,700 - £59,500)
Respiratory hospital admissions	Case of a hospital admission, of average duration 8 days	£9,800 (£3,300 – £16,300)	£3,500 – £14,500
Cardiovascular hospital admissions	Case of a hospital admission, of average duration 9 days	£10,000 (£3,500 – £16,500)	£4,100 – £13,500
QALY loss	Cumulative discounted QALYs over duration of disease	£72,000 (£36,000 - £96,100)	N/A

*Values rounded to nearest £100
**Values uplifted to 2022 using HMT deflators and adjustment made for income growth from 2012-22

4 Estimation and valuation of non-health impact pathways

4.1 Material damage and building soiling

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 £270 per tonne of SO_2 emitted (2022 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[8] (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 2μgm^{-3}) to gain the impact per tonne of NO_x or VOC emitted via this ozone pathway in 2022 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 £642 (2022 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

[8] http://www.externe.info/externe_d7/sites/default/files/vol2.pdf, p.300, http://www.externe.info/externe_d7/sites/default/files/vol7.pdf, p. 381, http://www.externe.info/externe_d7/sites/default/files/methup05a.pdf, p 109.

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

4.2.1 Valuation of pathways in 2020 update

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. In an attempt to start to fill 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, Mills, & Milne, 2014) titled 'Assessment of the Impacts of Air Pollution on Ecosystem Services – Gap Filling and Research Recommendations'. 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, Mills, & Milne, 2014) also provided direction on the rigour of the value estimate. To do so they scored each damage cost as either '## Robust', '# Acceptable' or '(#) Improvements desirable and not currently acceptable for policy appraisal'.

In the 2020 damage costs, all pathways scored either '## Robust' or '# Acceptable' were included in the updated damage costs. As part of this update, a targeted review of the ecosystem impacts was undertaken, in particular comparing the pathways covered and results to studies in the EU and US. When aggregating the effects of additional pollution, it was observed that the UK comes to a different overall position regarding the effect of additional N relative to the US and EU. In the UK, additional N produces a net benefit (for NH_3 – the net effect is a net cost for NOx), whereas in the US and EU analyses additional N has a detrimental effect on ecosystems.

In part, this results from the approach to estimating impacts in the UK. Jones et al attempted to monetise several ecosystem service pathways separately – i.e. provisioning services separate to regulating services separate to cultural services. Effects are quantified where evidence is available, but gaps remained in both the services captured and the ecosystems covered (e.g. carbon sequestration captures the impacts in woodlands only, whereas appreciation for biodiversity, which is represented by species diversity, is assessed in heathland, acid grassland, sand dune grassland and bogs). As such when impacts are aggregated as they were to produce the damage costs, the overall net impact presented is more driven by which services were able to be quantified, rather than an attempt to consciously represent the true overall impact.

After further consideration, for 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. Given this reflects a negative effect, although this is still only a partial estimation, the overall direction of impact is more in line with US and EU approaches and the direction of impact that would be expected should all individual pathways and ecosystem service effects be captured.

(Jones, Mills, & Milne, 2014) also provided uncertainty ranges around the valuation of each damage. 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).

4.2.2 Updates to valuation approach in 2023 update

4.2.2.1 Ozone impacts on crop production

Defra commissioned a report by (Jones, Mills, & Milne, 2014) that analysed the ozone impacts on wheat production. 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, subject to a 3.5% discount rate. Calculation of economic loss used the five-year average farm gate wheat value, centred on 2007 (£109/tonne).

The value transfer guidance has been updated to reflect a more recent five-year average farm gate wheat value estimate, centred on 2019. 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 2017-2021 (137, 2015=100) was taken from Defra's latest API (Defra, 2022). The 63% change in API was then used to adjust the five-year average farm gate wheat value, centred on 2007 to 2019 (£178/tonne). The ecosystem service impacts of crop production were increased by 63% 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 4-2.

Table 4-2 Ecosystem service value applied in updated damage costs (£2019 per tonne) - Ozone impacts on wheat production

Pollutant	Unit	Sensitivity	Crop production
O_3*	£/POD (2019 prices)	Central	164,080,339
		Low	*136,121,983*
		High	*194,128,964*

4.2.2.2 Nitrogen impacts on livestock production

The report by (Jones, Mills, & Milne, 2014) analysed the impact of nitrogen deposition on improved grassland. 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[9]. 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, The John Nix Farm Management Pocketbook, 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

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

Management pocketbook reports indicative fertiliser input prices in the UK, based on August 2010 spot prices (Redman, The John Nix Farm Management Pocketbook, 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, The John Nix Pocketbook for Farm Management: 49th Edition for 2019, 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 4-3.

Table 4-3 Ecosystem service value applied in updated damage costs (£2019 per tonne) - Nitrogen impacts on livestock production

Pollutant	Unit	Sensitivity	Livestock production
NO2	£/tonne (2019 prices)	Central	-£9.23
		Low	*-£5.87*
		High	*-£12.37*
NH_3	*£/tonne (2019 prices)*	Central	*-£308.33*
		Low	*-£195.63*
		High	*-£415.06*

4.2.2.3 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). In this update, 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, The John Nix Pocketbook for Farm Management: 49th Edition for 2019, 2019).

Table 4-4: Ecosystem service value applied in updated damage costs (£2019 per tonne) – Ozone impacts on livestock production

Pollutant	Unit	Sensitivity	Livestock production
O_3	£/ppb (2019 prices)	Central	-1,195,341.30
		Low	*-485,642.95*
		High	*-1,939,159.77*

4.2.2.4 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 £54.90/tonneCO_2e for 2014 was adopted in the 2020 damage cost update was sourced from (DECC, 2010). The latest non-traded carbon price from the Green Book supplementary guidance of £248/tonneCO_2e for 2020 was applied in this analysis to scale up the costs (BEIS, 2021b).

Table 4-5: Ecosystem service value applied in updated damage costs (£2020 per tonne) – Nitrogen and ozone impacts on CO_2 emissions

Pollutant	Unit	Sensitivity	CO_2 GHG
NO2	£/tonne (2020 prices)	Central	-243.93
		Low	*-102.99*
		High	*-424.63*
NH_3	*£/tonne (2020 prices)*	Central	*-5,723.88*
		Low	*-2,418.56*
		High	*-9,956.14*
O_3	*£/tonne (2020 prices)*	Central	*-25,929,326.05*
		Low	*-17,754,962.96*
		High	*-35,862,877.96*

Table 4-6: Ecosystem service value applied in updated damage costs (£2020 per tonne) – Nitrogen impacts on N2O emissions

Pollutant	Unit	Sensitivity	N_2O GHG
NO2	£/tonne (2020 prices)	Central	53.30
		Low	*28.01*
		High	*84.47*
NH_3	*£/tonne (2020 prices)*	Central	*1,528.66*
		Low	*809.05*
		High	*2,427.60*

Table 4-7: Ecosystem service value applied in updated damage costs (£2020 per tonne) – Sulphur impacts on CH4 emissions

Pollutant	Unit	Sensitivity	CH_4 GHG
SO_2	£/tonne (2020 prices)	Central	-23.94
		Low	*-7.23*
		High	*-42.91*

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, burning oil, biomass, LPG, peat, petroleum coke
Domestic	Inner conurbation, Urban big, Urban medium, Urban small, Rural	Gas, coal, burning 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 2023, 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 2019 emission factors, updating price base to 2022 and adopting a dynamic emissions factor for electricity consumption.

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 2023 updated set of air pollutant damage costs.

The 2023 damage costs were derived from air pollutant modelling that was calculated using emission estimates from the NAEI 2019. Hence emission factors were also derived from NAEI 2019.

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 2019 from NAEI 2019, split by vehicle type and fuel
2. Take UK total fuel consumption for road traffic in 2019 by vehicle type and fuel from NAEI 2019

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, burning oil, biomass, LPG, peat, and petroleum coke) in 2019 for all sectors
2. Take UK total fuel consumption (gas, coal, burning oil, biomass, LPG, peat, and petroleum coke) in 2019 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, burning oil, biomass, LPG, Peat, and Petroleum Coke) in 2019 for the domestic sector
2. Take UK total fuel consumption (gas, coal, burning oil, biomass, LPG, Peat, and Petroleum Coke) in 2019 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.

The approach to estimating the above activity costs is consistent with that used for the 2020 update. However, for the present (2023) update, an improved approach has been adopted to estimate activity costs associated with electricity consumption. Previously, a static, historic emissions factor was used for all energy types to estimate activity costs. Given the projected decarbonisation of the electricity grid (and also complementary reduction in the air pollutant emissions intensity of generation), this approach is not ideal, in particular as activity costs are deployed by policy analysts to assess the co-benefits of switching energy consumption away from fossil fuels to electricity.

Instead, for this update, a dynamic set of emissions factors for electricity consumption, which reduce over time reflecting projected trends in decarbonisation of the energy grid, have been adopted. This results in a declining activity cost to electricity consumption over time.

In line with BEIS appraisal guidance around the estimation of GHG impacts of changing electricity consumption, a set of long-run marginal emissions factors are used. These are different to the average grid emissions factors, and better represent the impacts of changing demand at the margins.

The activity costs for electricity consumption are estimated following these steps:

1. Take 2023 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 2019 emissions of NOx, $PM_{2.5}$ and SO_2 from NAEI 2019 for 'power stations'
3. Take final consumption (excluding international aviation) of electricity for 2019 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 2019.
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 CO2 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 $PM_{2.5}$ and 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.

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 no longer applied to define the activity costs in future years. This is consistent with broader changes made in the 2023 update to the damage costs and reflects latest 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 2019. The 2019 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 2019 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 2019			Total fuel consumption in 2019	Calculated emissions factor		
		$PM_{2.5}$	NOx	SO_2		$PM_{2.5}$	NOx	SO_2
		kTonnes			*Million litres*	*g/litre*		
Transport	Car petrol	0.327	17.111	0.129	14,566	0.0224	1.175	0.009
	Car diesel	2.381	128.013	0.157	13,048	0.1825	9.811	0.012
	LGV petrol	0.006	0.218	0.002	239	0.0232	0.911	0.009
	LGV diesel	1.098	99.264	0.088	7,313	0.1502	13.573	0.012
	Rigid HGV diesel	0.202	13.939	0.034	2,841	0.0712	4.906	0.012
	Articulated HGV diesel	0.152	7.756	0.056	4,701	0.0324	1.650	0.012
		kTonnes			*kWh*	*Tonnes/GWh*		
National	Gas	1.92	133.02	1.91	780,444,946,848	0.002	0.170	0.002
	Coal	7.41	26.82	40.54	38,679,183,079	0.191	0.693	1.048
	Burning oil	8.40	205.98	11.30	109,572,501,298	0.077	1.880	0.103
	Biomass	43.27	18.38	1.96	72,686,373,086	0.595	0.253	0.027
	LPG	0.04	2.89	0.03	12,292,299,912	0.003	0.235	0.002
	Peat	0.04	0.00	0.00	12,664,392	2.837	0.180	0.040
	Petroleum coke	0.72	6.85	54.67	13,831,276,962	0.052	0.495	3.953
Domestic	Gas	1.21	19.48	0.30	279,695,246,711	0.004	0.070	0.001
	Coal	3.52	2.73	12.32	5,833,511,212	0.603	0.467	2.113
	Burning oil	0.16	4.33	0.58	23,606,673,338	0.007	0.184	0.024
	Biomass	41.10	6.23	0.96	24,340,823,761	1.689	0.256	0.040
	LPG	0.01	0.55	0.00	3,009,062,578	0.004	0.184	0.001
	Peat	0.04	0.00	0.00	12,664,392	2.837	0.180	0.040
	Petroleum coke	0.40	0.94	29.31	1,945,260,649	0.203	0.482	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	2022	2025	2030	2035	2040	2045	2050
$PM_{2.5}$	0.005	0.004	0.002	0.001	0.000*	0.000*	0.000*
NOx	0.264	0.220	0.127	0.044	0.015	0.009	0.007
SO_2	0.041	0.034	0.020	0.007	0.002	0.001	0.001

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

5.2.3 Emissions to concentration modelling

Some additional PCM model runs were completed in order to derive the emissions to concentration relationships for domestic emissions in the required locations (i.e. 'area types') and for the relevant fuels. This entailed adapting the scripts and spreadsheets used to generate the damage costs for road transport by area type to generate emissions to concentrations relationships for domestic emissions by area type and fuel.

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 burning 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 Burning Oil Inner Conurbation	Domestic Burning Oil Urban Big	Domestic Burning Oil Urban Medium	Domestic Burning Oil Urban Small	Domestic Burning 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 2023 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 2020 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 addition to the new pathways discussed above, 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 chronic 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	Deaths brought forward	L	C	H
SO_2	Respiratory hospital admission	L	C	H
O_3	Deaths brought forward	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 acute deaths using the 'good health VOLY'

No range is recommended by the IGCB around the value of deaths 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 acute deaths 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 acute deaths 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 acute deaths being valued using this higher valuation in the high damage cost estimate.

6.2.3 Life-years-lost per acute death

In order to convert the number of deaths brought forward as a consequence of acute 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 acute effects 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 acute death).

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.

6.2.4 Unit impact values

As noted above in Table 3-6, uncertainty ranges around each health endpoint are applied and varied under the low and high damage cost range.

7 Updated damage costs

7.1 2023 results

The updated set of damage costs are presented in the following tables, alongside the low and high estimated sensitivities around the central values. These values represent the damage costs associated with pollutant emissions in 2022, presented in 2022 prices. All sustained impacts of 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 2022, impacts in 2023, 2024, 2025, etc are discounted back to 2022). A positive damage cost represents a cost associated with an increase in pollutant emissions or a benefit associated with a decrease in pollutants emissions.

Note that for these damage costs the change in $PM_{2.5}$ emission is the preferred metric for PM 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 2019. These are presented in Section 2.7.

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.

National damage costs are listed in Table 7-1. The damage costs for VOC include impacts via the O_3 pathways only. Sector specific damage costs for $PM_{2.5}$ and NO_x are provided in Table 7-2 and Table 7-3.

Table 7-4 disaggregates a selection of the damage costs by their contributing impact pathways, including the low and high sensitivity damage costs. It can be seen from this table that:

- 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, chronic exposure to PM on mortality is still an important effect (but in this case PM is a 'secondary' pollutant), but the mortality effect of chronic exposure to NO2 is the largest single pathway. This is the case even though the adjustment to account for the overlap between the two chronic 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, 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 2022:

- 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 first year of the appraisal period should be discounted using the Green Book 2022 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, 2022).

Table 7-1 – Revised national damage cost estimates and sensitivity bounds (2022 prices, impacts discounted to 2022). $PM_{2.5}$ is the preferred metric for the change in PM emissions

Pollutant Emitted	Central Damage Cost (£/t)	Low – High damage cost sensitivity range (£/t)	
		Low sensitivity damage cost	High sensitivity damage cost
NOx	8,148	1,567	30,282
SO_2	16,616	6,615	43,850
NH_3	9,667	3,727	26,172
VOC	172	104	309
$PM_{2.5}$	74,769	29,631	212,839

Table 7-2 – Revised sector PM damage cost estimates and sensitivity bounds (2022 prices, impacts discounted to 2022). $PM_{2.5}$ is the preferred metric for the change in PM emissions

Pollutant Emitted	Central Damage Cost (£/t)	Low – High damage cost sensitivity range (£/t)	
		Low sensitivity damage cost	High sensitivity damage cost
$PM_{2.5}$ Part A Category 1	8,583	3,386	23,231
$PM_{2.5}$ Part A Category 2	31,972	12,619	86,835
$PM_{2.5}$ Part A Category 3	155,496	61,402	424,674
$PM_{2.5}$ Part A Category 4	3,537	1,396	9,616
$PM_{2.5}$ Part A Category 5	6,712	2,655	18,716
$PM_{2.5}$ Part A Category 6	17,707	6,996	48,650
$PM_{2.5}$ Part A Category 7	1,581	624	4,307
$PM_{2.5}$ Part A Category 8	3,224	1,273	8,807
$PM_{2.5}$ Part A Category 9	7,834	3,110	22,721
$PM_{2.5}$ Industry (area)	76,354	30,677	251,116
$PM_{2.5}$ Commercial	59,509	23,426	156,656
$PM_{2.5}$ Domestic	84,629	33,307	222,144
$PM_{2.5}$ Solvents	106,415	42,166	302,369
$PM_{2.5}$ Road Transport	84,548	33,533	242,761
$PM_{2.5}$ Aircraft	76,064	30,015	206,048
$PM_{2.5}$ Offroad	53,014	20,860	138,750
$PM_{2.5}$ Rail	56,685	22,338	151,089
$PM_{2.5}$ Ships	24,027	9,460	63,399
$PM_{2.5}$ Waste	72,008	28,359	190,532
$PM_{2.5}$ Agriculture	28,654	11,946	129,361
$PM_{2.5}$ Other	85,253	33,594	227,104
$PM_{2.5}$ Road Transport Central London	472,656	187,305	1,344,541
$PM_{2.5}$ Road Transport Inner London	450,215	178,418	1,281,130
$PM_{2.5}$ Road Transport Outer London	246,942	97,937	708,775
$PM_{2.5}$ Road Transport Inner Conurbation	167,746	66,522	480,962
$PM_{2.5}$ Road Transport Outer Conurbation	104,833	41,578	301,005
$PM_{2.5}$ Road Transport Urban Big	96,592	38,306	277,068
$PM_{2.5}$ Road Transport Urban Large	78,835	31,261	225,880
$PM_{2.5}$ Road Transport Urban Medium	63,766	25,284	182,555
$PM_{2.5}$ Road Transport Urban Small	52,114	20,663	149,131
$PM_{2.5}$ Road Transport Rural	31,972	12,683	92,012
$PM_{2.5}$ Rail Transport Central London	428,863	168,846	1,130,506
$PM_{2.5}$ Rail Transport Inner London	421,032	165,762	1,109,872
$PM_{2.5}$ Rail Transport Outer London	238,024	93,712	627,528
$PM_{2.5}$ Rail Transport Inner Conurbation	145,530	57,296	383,629
$PM_{2.5}$ Rail Transport Outer Conurbation	77,259	30,421	203,985
$PM_{2.5}$ Rail Transport Urban Big	81,071	31,920	213,881
$PM_{2.5}$ Rail Transport Urban Large	78,163	30,793	207,664
$PM_{2.5}$ Rail Transport Urban Medium	52,508	20,679	138,921
$PM_{2.5}$ Rail Transport Urban Small	39,677	15,641	106,243
$PM_{2.5}$ Rail Transport Rural	26,771	10,555	71,762

Table 7-3 – Revised sector NOx national damage cost estimates and sensitivity bounds (2022 prices, impacts discounted to 2022).

Pollutant Emitted	Central Damage Cost (£/t)	Low – High damage cost sensitivity range (£/t)	
		Low sensitivity damage cost	High sensitivity damage cost
NOx Part A Category 1	3,103	1,029	9,377
NOx Part A Category 2	4,203	1,146	13,935
NOx Part A Category 3	8,467	1,602	31,603
NOx Part A Category 4	2,842	1,001	8,296
NOx Part A Category 5	3,155	1,034	9,591
NOx Part A Category 6	4,356	1,162	14,569
NOx Part A Category 7	2,757	992	7,942
NOx Part A Category 8	3,021	1,020	9,036
NOx Part A Category 9	3,307	1,050	10,222
NOx Industry (area)	8,635	1,619	32,298
NOx Commercial	16,583	2,469	65,232
NOx Domestic	12,881	2,073	49,893
NOx Solvents	14,796	2,278	57,829
NOx Road Transport	11,682	1,945	44,927
NOx Aircraft	11,268	1,901	43,208
NOx Offroad	7,881	1,539	29,175
NOx Rail	8,650	1,621	32,364
NOx Ships	3,877	1,111	12,584
NOx Waste	8,477	1,603	31,645
NOx Agriculture	3,810	1,104	12,306
NOx Other	3,678	1,090	11,759
NOx Road Transport Central London	63,051	7,433	257,783
NOx Road Transport Inner London	60,239	7,132	246,132
NOx Road Transport Outer London	33,064	4,229	133,527
NOx Road Transport Inner Conurbation	22,630	3,115	90,291
NOx Road Transport Outer Conurbation	14,408	2,236	56,220
NOx Road Transport Urban Big	13,341	2,122	51,800
NOx Road Transport Urban Large	11,013	1,874	42,151
NOx Road Transport Urban Medium	9,054	1,664	34,037
NOx Road Transport Urban Small	7,545	1,503	27,782
NOx Road Transport Rural	4,921	1,223	16,908
NOx Rail Transport Central London	56,456	6,729	230,452
NOx Rail Transport Inner London	56,808	6,767	231,905
NOx Rail Transport Outer London	33,029	4,228	133,370
NOx Rail Transport Inner Conurbation	20,580	2,899	81,784
NOx Rail Transport Outer Conurbation	11,902	1,973	45,820
NOx Rail Transport Urban Big	11,763	1,959	45,244
NOx Rail Transport Urban Large	10,649	1,841	40,624
NOx Rail Transport Urban Medium	7,817	1,539	28,886
NOx Rail Transport Urban Small	6,230	1,371	22,305
NOx Rail Transport Rural	4,554	1,192	15,357

Table 7-4 - Updated national damage costs for 2022 and contributing pathways (£2022 prices, impacts discounted to 2022) – Central

Pollutant Emitted	NOx	SO_2	NH_3	VOC	$PM_{2.5}$
Damage Cost (£/t)	**8,148**	**16,616**	**9,667**	**172**	**74,769**
$PM_{2.5}$ Mortality (long-term exposure)	1,565	9,321	5,275	0	42,392
$PM_{2.5}$ Respiratory hospital admission	10	57	32	0	259
$PM_{2.5}$ 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 Deaths brought forward	0	39	0	0	0
SO_2 Respiratory hospital admission	0	86	0	0	0
O_3 Deaths brought forward	-11	0	0	4	0
O_3 Respiratory hospital admission	-62	0	0	24	0
O_3 Cardiovascular hospital admission	0	0	0	0	0
NO_2 Respiratory hospital admission	85	0	0	0	0
NO_2 Cardiovascular hospital admission	0	0	0	0	0
NO_2 Deaths brought forward	0	0	0	0	0
NO_2 Mortality(long-term exposure)	2,692	0	0	0	0
$PM_{2.5}$ Productivity	94	557	315	0	2,535
PM_{10} Productivity	0	0	0	0	0
O_3 Productivity	-63	0	0	24	0
O_3 Productivity	0	0	0	0	0
NO_2 Productivity	0	0	0	0	0
O_3 Material damage	-20	0	0	6	0
PM_{10} Building soiling	0	0	0	0	994
SO_2 Material damage	0	270	0	0	0
SO_2 Ecosystems	0	0	0	0	0
O_3 Ecosystems	-72	0	0	45	0
O_3 Ecosystems	-32	0	0	68	0
NO_2 Ecosystems	121	0	0	0	0
NH_3 Ecosystems	0	0	487	0	0
PM_{10} Chronic Bronchitis Incidence	0	0	0	0	0
$PM_{2.5}$ IHD Incidence	153	910	515	0	4,138
NO_2 Asthma (Adults) Incidence	0	0	0	0	0
$PM_{2.5}$ Stroke Incidence	266	1,584	896	0	7,202
$PM_{2.5}$ Diabetes Incidence	0	0	0	0	0
NO_2 Diabetes Incidence	0	0	0	0	0
$PM_{2.5}$ Lung Cancer Incidence	20	121	69	0	552
NO_2 Lung Cancer Incidence	0	0	0	0	0
$PM_{2.5}$ Asthma (Children) Incidence	616	3,671	2,078	0	16,697
NO_2 Asthma (Small Children) Incidence	2,079	0	0	0	0
NO_2 Asthma (Older Children) Incidence	708	0	0	0	0

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

Table 7-5 Updated national damage costs for 2022 and contributing pathways (£2022 prices, impacts discounted to 2022) – Low

Pollutant Emitted	NOx	SO_2	NH_3	VOC	$PM_{2.5}$
Damage Cost (£/t)	**1,567**	**6,615**	**3,727**	**104**	**29,631**
$PM_{2.5}$ Mortality (long-term exposure)	888	5,291	2,994	0	24,064
$PM_{2.5}$ Respiratory hospital admission	-2	-13	-7	0	-57
$PM_{2.5}$ 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 Deaths brought forward	0	20	0	0	0
SO_2 Respiratory hospital admission	0	29	0	0	0
O_3 Deaths brought forward	-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	17	0	0	0	0
NO_2 Cardiovascular hospital admission	0	0	0	0	0
NO_2 Deaths brought forward	0	0	0	0	0
NO_2 Mortality(long-term exposure)	443	0	0	0	0
$PM_{2.5}$ Productivity	32	188	107	0	856
PM_{10} Productivity	0	0	0	0	0
O_3 Productivity	-19	0	0	7	0
O_3 Productivity	0	0	0	0	0
NO_2 Productivity	0	0	0	0	0
O_3 Material damage	-20	0	0	6	0
PM_{10} Building soiling	0	0	0	0	994
SO_2 Material damage	0	270	0	0	0
SO_2 Ecosystems	0	0	0	0	0
O_3 Ecosystems	-47	0	0	30	0
O_3 Ecosystems	-26	0	0	57	0
NO_2 Ecosystems	39	0	0	0	0
NH_3 Ecosystems	0	0	164	0	0
PM_{10} Chronic Bronchitis Incidence	0	0	0	0	0
$PM_{2.5}$ IHD Incidence	-11	-65	-37	0	-296
NO_2 Asthma (Adults) Incidence	0	0	0	0	0
$PM_{2.5}$ Stroke Incidence	-12	-72	-41	0	-327
$PM_{2.5}$ Diabetes Incidence	0	0	0	0	0
NO_2 Diabetes Incidence	0	0	0	0	0
$PM_{2.5}$ Lung Cancer Incidence	5	27	15	0	123
NO_2 Lung Cancer Incidence	0	0	0	0	0
$PM_{2.5}$ Asthma (Children) Incidence	158	940	532	0	4,275
NO_2 Asthma (Small Children) Incidence	135	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 2022 and contributing pathways (£2022 prices, impacts discounted to 2022) – High

Pollutant Emitted	NOx	SO_2	NH_3	VOC	$PM_{2.5}$
Damage Cost (£/t)	**30,282**	**43,850**	**26,172**	**309**	**212,839**
$PM_{2.5}$ Mortality (long-term exposure)	2,173	12,940	7,323	0	58,852
$PM_{2.5}$ Respiratory hospital admission	43	254	144	0	1,155
$PM_{2.5}$ Cardiovascular hospital admission	8	50	28	0	226
PM_{10} Respiratory hospital admission	0	0	0	0	0
PM_{10} Cardiovascular hospital admission	0	0	0	0	0
SO_2 Deaths brought forward	0	67	0	0	0
SO_2 Respiratory hospital admission	0	143	0	0	0
O_3 Deaths brought forward	-31	0	0	12	0
O_3 Respiratory hospital admission	-166	0	0	65	0
O_3 Cardiovascular hospital admission	-9	0	0	3	0
NO_2 Respiratory hospital admission	203	0	0	0	0
NO_2 Cardiovascular hospital admission	91	0	0	0	0
NO_2 Deaths brought forward	0	0	0	0	0
NO_2 Mortality (long-term exposure)	7,347	0	0	0	0
$PM_{2.5}$ Productivity	207	1,235	699	0	5,619
PM_{10} Productivity	12	61	38	0	391
O_3 Productivity	-202	0	0	79	0
O_3 Productivity	-4	0	0	2	0
NO_2 Productivity	0	0	0	0	0
O_3 Material damage	-20	0	0	6	0
PM_{10} Building soiling	0	0	0	0	994
SO_2 Material damage	0	270	0	0	0
SO_2 Ecosystems	0	0	0	0	0
O_3 Ecosystems	-100	0	0	63	0
O_3 Ecosystems	-37	0	0	81	0
NO_2 Ecosystems	279	0	0	0	0
NH_3 Ecosystems	0	0	1,202	0	0
PM_{10} Chronic Bronchitis Incidence	1,548	7,815	4,845	0	50,019
$PM_{2.5}$ IHD Incidence	466	2,773	1,569	0	12,611
NO_2 Asthma (Adults) Incidence	3,305	0	0	0	0
$PM_{2.5}$ Stroke Incidence	806	4,799	2,716	0	21,826
$PM_{2.5}$ Diabetes Incidence	813	4,845	2,742	0	22,035
NO_2 Diabetes Incidence	6,083	0	0	0	0
$PM_{2.5}$ Lung Cancer Incidence	42	252	142	0	1,145
NO_2 Lung Cancer Incidence	90	0	0	0	0
$PM_{2.5}$ Asthma (Children) Incidence	1,402	8,348	4,724	0	37,966
NO_2 Asthma (Small Children) Incidence	4,075	0	0	0	0
NO_2 Asthma (Older Children) Incidence	1,857	0	0	0	0

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

7.2 Comparison to previous estimates

For comparison, the updated central damage costs are presented alongside the original set of costs and those published in 2020, 2019 and 2015 in Table 7-7 and Table 7-8. The original central damage cost estimates have also been included in the tables to provide a more direct comparison between the updated and original damage costs (uplifted to 2022 prices in Table 7-8 to remove the impact of changing price base).

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)
NOx National	875	*	6,199	6,385	8,148
NOx Domestic	*	14,646	13,950	12,448	12,881
NOx Industry***	*	13,131	*	*	*
NOx Industry (area sources) ***	*	*	5,671	5,891	8,635
NOx Road Transport	*	25,252	10,699	9,066	11,682
SO_2	1,496	1,956	6,273	13,026	16,616
NH_3	1,884	2,363	6,046	7,923	9,667
VOC	*	*	102	102	172
$PM_{2.5}$	*	*	105,836	73,403	74,769
$PM_{2.5}$ Domestic	25,770	33,713	85,753	89,456	84,629
$PM_{2.5}$ Industry ***	23,103	30,225	*	*	*
$PM_{2.5}$ Industry (area sources) ***	*	*	95,847	71,455	76,354
$PM_{2.5}$ Road Transport	44,430	58,125	203,331	81,518	84,548
$PM_{2.5}$ Waste	19,105	24,994	162,082	74,029	72,008

Table 7-8 - Updated and original central damage cost estimates uplifted to 2022 prices

Pollutant	Original damage cost (£2022/t)	Damage costs 2015 (£2022/t)	Damage costs 2019 (£2022/t)	Damage costs 2020 (£2022/t)	Damage costs 2023 (£2022/t)
NOx National	1,248	*	6,992	7,202	8,148
NOx Domestic	*	17,140	15,736	14,041	12,881
NOx Industry***	*	15,367	*	*	*
NOx Industry (area sources) ***	*	*	6,397	6,645	8,635
NOx Road Transport	*	29,553	12,068	10,226	11,682
SO_2	2,133	2,289	7,076	14,693	16,616
NH_3	2,687	2,765	6,820	8,937	9,667
VOC	*	*	115	115	172
$PM_{2.5}$	*	*	119,383	82,798	74,769
$PM_{2.5}$ Domestic	36,751	39,455	96,729	100,906	84,629
$PM_{2.5}$ Industry ***	32,948	35,373	*	*	*
$PM_{2.5}$ Industry (area sources) ***	*	*	108,115	80,601	76,354
$PM_{2.5}$ Road Transport	63,363	68,025	229,357	91,952	84,548
$PM_{2.5}$ Waste	27,246	29,251	182,828	83,505	72,008

* = 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.

7.2.1 How do the damage costs compare?

As can be seen from Table 7-7, the updated damage costs show variance from both the original and latest published sets of damage costs. The changes differ by damage cost.

For NOx:

- The 2023 damage costs have increased by 13% compared to the 2020 damage cost (expressed in 2022 prices).
- A key driver is the increase in the CRF for mortality associated with long-term exposure to $PM_{2.5}$ and an upward revision of the ugm^{-3} change in secondary inorganic aerosol (SIA) per tonne of

NOx, resulting in an increase in QALY loss assigned. Updating the QALY loss value has also marginally increased the damage costs.

For SO_2:

- The 2023 damage costs have increased by 13% compared to the 2020 damage cost (expressed in 2022 prices).
- This is a result from an increase in the CRF used for mortality associated with long-term exposure to $PM_{2.5}$ (this effect also influences all other damage costs). Updating the QALY loss value has also marginally increased the damage costs.

For NH_3:

- The damage cost is now 8% higher than the 2020 update (expressed in 2022 prices).
- Key drivers here are again the increase in the CRF for mortality associated with long-term exposure $PM_{2.5}$ and updating the QALY loss value. Furthermore, the damage cost for ammonia has also increased due to the change to select only 'robust' ecosystem pathways.

For VOC:

- The damage cost for VOC was first included in the 2019 damage costs. The value in the 2023 damage costs has almost increased by 50%.
- The increase is related to updating ecosystem valuations for ozone impacts on livestock production and ozone impacts of CO_2 sequestration to reflect the latest feed prices and carbon price.

For $PM_{2.5}$:

- The damage costs increased significantly between the 2015 and the 2019 update to the damage costs, as additional morbidity pathways based on the PHE model were added. A discussion of the differences between the 2019 and earlier estimates can be found in (Birchby D. , Stedman, Whiting, & Vedrenne, 2019).
- In the present 2023 update, the $PM_{2.5}$ damage cost has decreased by 10% relative to the central 2020 set.

  The key drivers of this change have been the updates to the CRF for incidence of IHD (which is associated with a lower CRF). Likewise, the CRF for stroke has also been revised downwards.

  The changes above which have driven an overall reduction in the damage cost, have been partly (but not wholly) offset by the increase in the CRF for mortality associated with long-term exposure to $PM_{2.5}$ and update to the QALY value.

7.2.2 Differences between the 2020 and 2023 damage costs

There have been a number of updates to the damage costs in 2023. Most of the methodological changes have an upward effect on all the damage costs. However, with the $PM_{2.5}$ damage cost, the update to CRFs for two morbidity effects associated with chronic exposure has had a simultaneous downward effect, more than offsetting the other aforementioned updates, causing an overall drop.

In addition to the standard updates to the dispersion and emission modelling (to ensure that damage costs reflect the most recent air pollution levels), and the underlying population and baseline health data, the 2023 damage cost update have also included the following specific changes explained at a:

1. <u>Update several concentration response functions (CRFs) to reflect latest COMEAP guidance (this included updating the link between mortality and chronic exposure to $PM_{2.5}$)</u>

Following clear recommendations from COMEAP, for all damage costs the CRFs for mortality effects, associated with chronic exposure to $PM_{2.5}$ has been increased in line with WHO's systematic review. As a result, this has increased the value of this important pathway for all damage costs (except for VOCs) by 33% in the central case. As noted above, the CRF linking stroke and IHD incidence with chronic exposure have also been revised, in this case downward, in turn having a downward effect on the damage costs.

2. Aligning with the updated HMT Green Book, namely in relation to discounting and value of a QALY

To reflect the latest HMT Green Book guidance (2022)[10], the damage cost modelling has been updated to apply a discount rate for human health effects of 1.5% (the health specific discount rate), replacing the 3.5% discount rate and 2% per annum uplift used previously. To note, this is a small technical change made purely to align with HMT guidance and has no effect on the overall valuation. Moreover, we made this revision in agreement with the Defra central analysis team's steer.

Also, the QALY value used to monetise the chronic morbidity pathways has been uplifted slightly to reflect the latest Green Book guidance.

3. Adding new rail damage costs split by area type (to reflect DfT stakeholder needs)

This represents a fairly minor change which involves adding a new section to the damage costs publication to reflect DfT stakeholder needs. It is worth noting that these rail damage costs have been produced previously just not published.

4. Update to ecosystem pathways

The Defra commissioned Jones et al. (2014)[11] report 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, the report also scored each damage cost as either '## Robust', '# Acceptable' or '(#) Improvements desirable and not currently acceptable for policy appraisal'. In the 2020 damage costs, all pathways which scored either '## Robust' or '# Acceptable' were included in the updated damage costs. Following further review and consideration, in particular following the minor changes made to the valuation of some of these pathways (e.g. to reflect updated feed and carbon prices), for the 2023 update, only the 'robust' pathways are included in the damage costs; and so the 'acceptable' pathways have been removed. This change produced an 11% increase in the NH_3 damage cost, a 32% increase in the VOC damage cost, but no or negligible (i.e. <0.1%) change in all other DCs.

7.3 Damage costs per unit of concentration

Damage costs are typically expressed per tonne of emission. However, damage costs can also be expressed per unit of concentration, and damage costs expressed in this way are being increasingly used in policy appraisal. Generating these damage costs essentially involved removing the first step of the impact pathway – the relationship between 1 tonne of emission to a varying concentration impact. Instead, damages are calculated based on a 1 ugm^{-3} change in concentration.

The ability to apply such damage costs depends on the availability of more detailed concentration modelling. However, where such modelling is available, deploying damage costs per unit of 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).

Given the increasing use of these values in appraisal, this update also separately presents a set of damage costs per unit of concentration change. These are consistent with the damage costs per tonne estimated above. These values are presented in the table below.

[10] HMT (2022) The Green Book: appraisal and evaluation in central government. The 2022 version of the guidance can be found here.
[11] Jones et al. (2014) Assessment of the Impacts of Air Pollution on Ecosystem Services – Gap Filling and Research Recommendations (Defra Project AQ0827). The final report can be found here.

Table 7-9 – Damage costs per concentration change – Central damage cost sensitivity

Primary Pollutant	Sector	Direct impacts - (£2022 per population-weighted mean 1 μgm^{-3} change per person)	Other impacts (£2022 / tonne)
NO_X	National	7.30	2,585
SO_2	National	0.14	16,490
NH_3	National	-	9,667
VOC	National	-	172
$PM_{2.5}$	National	54.18	-

Notes on using the damage costs per change in concentration:

The damage costs per change in concentration are developed in a specific way, and hence need to be used following specific steps. In particular:

- Damage costs are expressed per 1 μgm^{-3} change in population-weighted concentrations – 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 damage 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 damage cost.
- 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. These include: impacts on productivity, ecosystems and materials. As such, where the damage costs per change in concentration are deployed to estimate effects, analysts should still apply the ‘other impacts’ expressed per tonne to the underlying change in tonnes of emissions. These two estimates are then added together to generate the total damage estimate.

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 2019 mapped emissions inventory. It is not possible to calculate an activity costs for these fuels in these areas. This is indicated as "N/A" in Table A5.

Where activity costs are deployed to assess impacts in years after 2022:

- 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 2022 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, 2022).

Table 8-1 – Transport activity costs (p/litre; impacts in 2022 in £2022 prices; update to BEIS's 'Table 14')

Area	Car petrol	Car diesel	LGV petrol	LGV diesel	Rigid HGV diesel	Articulated HGV diesel
Transport average	1.58	13.02	1.28	17.15	6.35	2.22
Central London	8.48	70.50	6.85	92.70	34.32	11.96
Inner London	8.10	67.33	6.55	88.54	32.78	11.42
Outer London	4.45	36.96	3.60	48.61	18.00	6.28
Inner conurbation	3.05	25.28	2.47	33.25	12.32	4.30
Outer conurbation	1.94	16.07	1.57	21.15	7.83	2.74
Urban big	1.80	14.87	1.45	19.58	7.25	2.53
Urban large	1.49	12.26	1.20	16.15	5.98	2.09
Urban medium	1.22	10.07	0.99	13.27	4.92	1.72
Urban small	1.02	8.37	0.82	11.04	4.09	1.43
Transport rural	0.66	5.43	0.54	7.18	2.66	0.94

Table 8-2 – National and domestic activity costs (p/kWh; impacts in £2022 prices; update to BEIS's 'Table 15')

Sector	Fuel	2022	2023	2024	2025	2030	2035	2040	2045	2050
NATIONAL AVERAGE	Electricity	0.155	0.147	0.138	0.129	0.075	0.026	0.009	0.006	0.004
	Gas	0.161	0.161	0.161	0.161	0.161	0.161	0.161	0.161	0.161
	Coal	3.738	3.738	3.738	3.738	3.738	3.738	3.738	3.738	3.738
	Burning oil	2.276	2.276	2.276	2.276	2.276	2.276	2.276	2.276	2.276
	Biomass	4.702	4.702	4.702	4.702	4.702	4.702	4.702	4.702	4.702
	LPG	0.217	0.217	0.217	0.217	0.217	0.217	0.217	0.217	0.217
	Peat	21.426	21.426	21.426	21.426	21.426	21.426	21.426	21.426	21.426
	Petroleum coke	7.364	7.364	7.364	7.364	7.364	7.364	7.364	7.364	7.364
DOMESTIC: Inner conurbation	Gas	0.231	0.231	0.231	0.231	0.231	0.231	0.231	0.231	0.231
	Coal	14.556	14.556	14.556	14.556	14.556	14.556	14.556	14.556	14.556
	Burning oil	0.632	0.632	0.632	0.632	0.632	0.632	0.632	0.632	0.632
	Biomass	29.199	29.199	29.199	29.199	29.199	29.199	29.199	29.199	29.199
	LPG	N/A	N/A	N/A	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	N/A	N/A	N/A
	Petroleum coke	29.567	29.567	29.567	29.567	29.567	29.567	29.567	29.567	29.567
DOMESTIC: Urban big	Gas	0.143	0.143	0.143	0.143	0.143	0.143	0.143	0.143	0.143
	Coal	9.617	9.617	9.617	9.617	9.617	9.617	9.617	9.617	9.617
	Burning oil	0.296	0.296	0.296	0.296	0.296	0.296	0.296	0.296	0.296
	Biomass	16.767	16.767	16.767	16.767	16.767	16.767	16.767	16.767	16.767
	LPG	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	Peat	27.937	27.937	27.937	27.937	27.937	27.937	27.937	27.937	27.937
	Petroleum coke	27.653	27.653	27.653	27.653	27.653	27.653	27.653	27.653	27.653
DOMESTIC: Urban medium	Gas	0.095	0.095	0.095	0.095	0.095	0.095	0.095	0.095	0.095
	Coal	7.542	7.542	7.542	7.542	7.542	7.542	7.542	7.542	7.542
	Burning oil	0.205	0.205	0.205	0.205	0.205	0.205	0.205	0.205	0.205
	Biomass	11.060	11.060	11.060	11.060	11.060	11.060	11.060	11.060	11.060
	LPG	N/A	N/A	N/A	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	N/A	N/A	N/A
	Petroleum coke	26.761	26.761	26.761	26.761	26.761	26.761	26.761	26.761	26.761
DOMESTIC: Urban small	Gas	0.076	0.076	0.076	0.076	0.076	0.076	0.076	0.076	0.076
	Coal	6.669	6.669	6.669	6.669	6.669	6.669	6.669	6.669	6.669
	Burning oil	0.170	0.170	0.170	0.170	0.170	0.170	0.170	0.170	0.170
	Biomass	8.740	8.740	8.740	8.740	8.740	8.740	8.740	8.740	8.740
	LPG	N/A	N/A	N/A	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	N/A	N/A	N/A
	Petroleum coke	26.409	26.409	26.409	26.409	26.409	26.409	26.409	26.409	26.409
DOMESTIC: Rural	Gas	0.060	0.060	0.060	0.060	0.060	0.060	0.060	0.060	0.060
	Coal	5.334	5.334	5.334	5.334	5.334	5.334	5.334	5.334	5.334
	Burning oil	0.129	0.129	0.129	0.129	0.129	0.129	0.129	0.129	0.129
	Biomass	5.540	5.540	5.540	5.540	5.540	5.540	5.540	5.540	5.540
	LPG	0.076	0.076	0.076	0.076	0.076	0.076	0.076	0.076	0.076
	Peat	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
	Petroleum coke	25.912	25.912	25.912	25.912	25.912	25.912	25.912	25.912	25.912

Appendix 1 – References

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