Benefit assessment methodologies for the LCP BREF implementation

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Summary

The methods for quantification of the damage caused by air pollution have been developed since 1990 and are now widely agreed both within the EU and outside. The full approach, the impact pathway approach (IPA), takes a logical and sequential pathway from emission, to exposure, to impact quantification and monetisation, and should be applied where costs on either side of the cost-benefit equation are likely to be substantial, and resources for a thorough analysis permit. A thorough analysis requires that emissions are tracked over extended distances (the whole of the EU and bordering countries) and account is taken of the formation of secondary pollutants, especially nitrate and sulphate aerosols, and ozone. It should also extend to the complete duration covered by the application. Arbitrary truncation of analysis in support of an application for derogation is to be avoided, as it will leave unclear the question of whether the benefits of allowing the derogation exceed the costs to society arising from additional health and other impacts. Applicants that undertake a full IPA assessment should provide a substantial amount of methodological information to support their case, documenting:

- Methods
- The geographic and temporal range of analysis
- Impacts quantified
- Impacts omitted from the analysis
- Data inputs
- Assumptions
- Results
- Treatment of uncertainty in comparing the costs and benefits of the application

An alternative approach applied by the European Environment Agency in quantification of the damage associated with individual power stations and other industrial facilities is to apply estimates of marginal damage costs expressed as €/tonne of emission. These have been derived using the full IPA, and are provided by the EEA as national averages. Some refinements to their use can be made by applying correction factors that take account of stack height and other relevant factors. Where it is not possible to complete a full application of the IPA using state of the art models, the €/tonne estimates provide a robust basis for comparing the costs and benefits of allowing or refusing a derogation.

Recognising the complexity of the IPA, applicants that use the full methodology should provide an estimate of the cost per tonne emission for each pollutant, linked to their analysis. This will permit data to be cross-checked with the EEA estimates and promote more structured and informed debate on the merits of permitting or refusing a derogation.

In addition to general discussion of methods this paper also provides information on updating damage cost estimates to ensure that the comparison of costs and benefits is valid, reflecting information on inflation over the last 2 decades, future economic growth and other factors.

It is noted that the assessment of damage costs, using the methods adopted by the European Commission is conservative for a number of reasons, particularly:

- It is not possible to include all pollution impacts
- The values adopted for mortality, in particular, are low compared to those recommended elsewhere (e.g. by OECD and USEPA).

These biases need to be considered in evaluation of any application for derogation.

Contents

1 Introduction

1.1 Objectives of this paper

The objective of this paper is to help ensure that Frank Bold Society and partner NGOs provide accurate information on estimating benefits as well as costs from retrofits of coal-fired power plants in the European Union in order to inform implementation processes of the LCP BREF at the national and local level. The LCP BREF sets new binding requirements for air pollutant emissions from large combustion plants but also foresees the possibility of derogation in some cases if disproportionality of retrofit costs in comparison to benefits can be demonstrated.

Within the paper a great deal of emphasis is placed on the damage costs per tonne emission reported by the European Environment Agency [1]. The EEA estimates, whilst not perfect, are discussed at various points within the paper, and have been calculated using state-of-the-art techniques.

1.2 Legislation

Applications for derogation under the Industrial Emissions Directive[2] (IED) are required under Article 15(4) to demonstrate that

"the achievement of emission levels associated with the best available techniques as described in BAT conclusions ***would lead to disproportionately higher costs compared to the environmental benefits*** *due to: (a) the geographical location or the local environmental conditions of the installation concerned; or (b) the technical characteristics of the installation concerned. The competent authority shall document in an annex to the permit conditions the reasons for the application of the first subparagraph including the result of the assessment and the justification for the conditions imposed. The emission limit values set in accordance with the first subparagraph shall, however, not exceed the emission limit values set out in the Annexes to this Directive, where applicable. The competent authority shall in any case ensure that no significant pollution is caused and that a high level of protection of the environment as a whole is achieved."*

The term 'significant pollution' is not defined in the IED. One indicator of significance concerns compliance with other legislation, for example:

1. Exceedance of ambient air or water quality standards, noting the requirements of Article 18 of the IED: "*Where an environmental quality standard requires stricter conditions than those achievable by the use of the best available techniques, additional measures shall be included in the permit, without prejudice to other measures which may be taken to comply with environmental quality standards*."
2. The requirements of legislation on natural ecosystems, for example through the Habitats Directive [3]. However, this Directive, like the IED, also contains reference to 'significant effects', without defining significance.

[1] https://www.eea.europa.eu/publications/costs-of-air-pollution-2008-2012
[2] http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32010L0075
[3] http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:31992L0043

Whilst this provides an indicator against which significance could be established, comparison with standards is insufficient: Environmental Directives follow a general guiding principle that environmental status "*should be maintained where it is already good, or improved*", with 'good' here interpreted as being within the legislated standards. To the extent that impacts are expected to occur instantaneously with exposure, maintenance of existing pollution loads might be expected as not impacting on environmental status. However, it is accepted that many impacts on health and natural ecosystems are a result of long-term exposures, with levels of pollution building up in the environment and the human body over time (good examples concern releases of toxic metals, emissions of oxidised or reduced nitrogen, and releases of greenhouse gases). On this basis, the maintenance of current levels of pollution is not necessarily equivalent to the maintenance of good environmental status.

If some level of harm beyond that linked to the implementation of BAT is to be accepted it is necessary to quantify impacts in order to describe 'significance'. Art. 15(4) of the IED states that:
"the Commission may, where necessary, assess and further clarify, through guidance, the criteria to be taken into account".

However, this guidance has not yet been developed[4], and it is uncertain if it will eventually be prepared by the Commission.

1.3 Methods

Following from the methods used in the development of European air quality legislation, methods should be based on the 'Impact Pathway Approach' (IPA) as illustrated in Figure 1 or the use of marginal damage costs derived using the IPA. The IPA was developed during the 1990s under the EU's ExternE (externalities of energy) research series [5]. It follows a logical and sequential pathway from emission to quantification of impacts and monetisation, and has now been adopted globally, as it demonstrates clear linkage between pollutant releases and impacts. It has been applied at all scales, from assessment for individual facilities, to assessment at the continental and global scales.

[4] A confidential report has been prepared for DG Environment by Ricardo Energy & Environment on methodologies for estimating BAT-related compliance costs and benefits. It evaluates the impact pathway approach and the marginal damage cost approach as well as the use of correction factors, and a fourth methodology based on avoided alternative abatement technology costs but is not discussed in this briefing.

[5] http://www.externe.info/externe_d7/

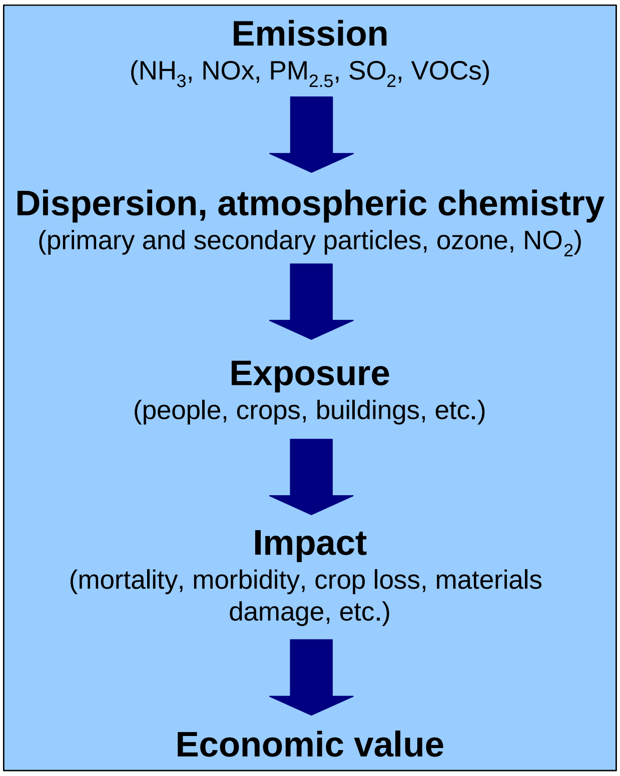

Figure 1. Illustration of the Impact Pathway Approach developed in the ExternE research studies and used in analysis for the European Commission

Whilst Figure 1 refers to a specific set of pollutants (NH_3, NOx, $PM_{2.5}$, SO_2 and VOCs), the approach can be used for any burden on health or the environment. Although the precise form of analysis will vary between pollutants [6], the overall format of following pollution from release, through atmospheric dispersion including transformation through chemical reactions where relevant, to exposure of the population, quantification of impact and valuation of impact, remains the same. Examples of its use include:

- The Impact Assessment developed by the European Commission for the IED draft in 2007 [7]
- The EU's Thematic Strategy on Air Pollution (TSAP) in 2005 under the Clean Air For Europe (CAFE) programme [8]
- The 2013 update to the TSAP (the draft Clean Air Policy Package) [9]

The IPA has also been used to generate the marginal damage (economic cost per tonne) adopted by the EEA for assessment of industrial pollution [10]. These estimates, provided at the national level, are an update to the damage costs provided in the guidance from the BREF on Economics and cross media effects [11], taking account of a number of updates in methods and data relating to:

- Dispersion modelling from EMEP [12]

[6] For toxic metals, for example, it is necessary to consider dietary intake as well as inhalation.
[7] http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A52007SC1679
[8] http://ec.europa.eu/environment/archives/cafe/
[9] http://ec.europa.eu/environment/air/pdf/TSAP%20CBA.pdf
[10] https://www.eea.europa.eu/publications/costs-of-air-pollution-2008-2012
[11] Annex 12 of http://eippcb.jrc.ec.europa.eu/reference/ecm.html.
[12] http://www.emep.int/index_model.html

- Response functions recommended by WHO under the Health Response to Air Pollutants in Europe (HRAPIE) study [13]
- Valuations adopted in the EC's benefits assessment for revision of the TSAP [9].

It is understood that the earlier figures have been used in derogation applications made recently in Poland: they no longer represent state of the art and should be replaced.

Of the two approaches, full implementation of the IPA should give the more robust result, as it can take specific account of stack height, the location of emission releases relative to sensitive receptors (people and ecosystems) and meteorology. However, it is more data intensive and complex to implement. If the full IPA is adopted, full documentation of the following factors is necessary to demonstrate the validity of estimated impacts and damage:

- Methods
- Range of analysis
- Impacts quantified
- Impacts omitted from the analysis
- Data inputs
- Assumptions
- Results
- Treatment of uncertainty in comparing the costs and benefits of the application

These issues are discussed in more depth in the following sections. Particular emphasis is given to the need to account for impacts over long distances (the whole of Europe) and to account for secondary as well as primary pollutants.

The application of damage per tonne factors provides a reasonable approximation and is the preferred option where a thorough application of the IPA is not possible. Given the long distances over which damage accumulates around a plant, concern that these factors do not adequately account for specific local conditions should be considered of little importance. Analysis based on local dispersion models, or that excludes atmospheric chemistry, will substantially underestimate damage costs. Local models are certainly useful for demonstrating compliance with air quality limit values, but this is not sufficient to demonstrate that a derogation should be granted, given that these limit values do not represent no-effect thresholds.

[13] http://www.euro.who.int/en/health-topics/environment-and-health/air-quality/publications/2013/health-risks-of-air-pollution-in-europe-hrapie-project.-recommendations-for-concentrationresponse-functions-for-costbenefit-analysis-of-particulate-matter,-ozone-and-nitrogen-dioxide

2 The scope of benefits to be considered in cost-benefit assessments of industrial air pollution

The EU has adopted the 'polluter pays principle' [14] as an overarching principle of environmental responsibility, requiring the internalisation of external costs. It then follows that all types of impact should be considered, wherever they occur. This section considers what this means in practice.

2.1 Pollutants

Analysis should clearly account for emissions of any pollutant whose emissions would be affected by the derogation applied for, and for which an emission limit value is set under the IED or for which environmental quality standards exist. This therefore includes the major air pollutants (NH_3, NOx, SO_2, $PM_{2.5}$) and various toxic metals and organic compounds.

Where emissions concern reactive pollutants (VOCs, NOx, SO_2, NH_3, etc.), full quantification of the damage also needs to account for impacts linked to secondary pollutants. The secondary pollutants of most interest are ozone (formed through reaction of NOx and VOCs) and ammonium sulphate and nitrate particles (formed through reaction of SO_2, NOx and NH_3).

For some facilities, the formation of secondary organic particles will also be of interest, though this should not apply to those where emissions are dominated by combustion.

Modelling of reactive pollutants that does not account for formation of secondary organic and inorganic particles, and ozone, will substantially underestimate damage associated with a facility: The unit damage costs published by the EEA are dominated by effects of exposure to these secondary pollutants for NH_3, NO_2, SO_2 and VOCs.

The damage associated with secondary pollutants (sulphate and nitrate particles and ozone) is attributed to whatever primary pollutant they originate from in the EEA's damage costs [10]. Hence the damage linked to human exposure to ammonium sulphate particles is not attributed to $PM_{2.5}$ but to SO_2 and NH_3.

2.2 Types of impact and economic value

A wide range of impacts can be associated with the pollutants of interest here. The damage costs published by the EEA (2014) includes those impacts shown in the list in italics.

- Human health
  - *Primary pollutants including fine particles, NO_2, SO_2 and associated secondary pollutants*
  - *Ozone precursors (VOCs and NOx)*
  - *Toxic metals and organics*
- Natural ecosystems
  - Nutrients such as NOx that cause eutrophication
  - Acidifying pollutants

[14] http://ec.europa.eu/environment/legal/law/pdf/principles/2%20Polluter%20Pays%20Principle_revised.pdf

- Ozone precursors (VOCs and NOx)
- Ammonia, chiefly from agriculture

- Crops
  - *Ozone precursors (VOCs and NOx)*
  - *Toxic metals and organics (via human uptake)*
- Forests
  - Ozone precursors (VOCs and NOx)
- Materials
  - *Acidifying pollutants*
  - Ozone precursors (VOCs and NOx)
- Visibility
  - Particles, NO_2, SO_2 and reaction products

The list is of course a simplification, as a full listing would identify the many different types of health impact associated with each pollutant [15].

Further work permits quantification of some of the missing effects since the EEA analysis was undertaken [16].

Quantified effects of air pollution (in general, and for power plants) are dominated by impacts on mortality - especially due to chronic mortality [17] from $PM_{2.5}$ concentrations. Depending on the approach used to value mortality, these effects account for between roughly 70% and 95% of total monetary impacts.

Health impacts should be costed in terms of:

- Health care costs
- Lost productivity
- Lost utility

All elements are included in the EEA's damage costs, to the extent that quantification is possible. The term 'utility' concerns what we value in 'health', a long life expectancy, freedom from pain and suffering, the ability to undertake physical activity, ranging from basic self care to vigorous exercise.

Impacts on crops are typically quantified in terms of lost production, whilst impacts on forests have been quantified against effects on production and carbon sequestration [16].

Impacts on materials have been quantified for acidifying pollutants and for particulate soiling, though only for 'utilitarian' buildings, accounting for repair costs. Much concern focused on impacts on cultural heritage during the acid rain debate of the 1970s, 80s and 90s, though such damage has not been integrated in economic estimates. This problem has declined in much of Europe, in line with major falls in emissions of SO_2 which was the most aggressive pollutant (in the UK, for example, SO_2 emissions are down by about 95% from the peak). The problem, may still be significant in parts of Europe where SO_2 levels are still elevated significantly.

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[15] For illustration see the 2016 report from the UK Royal Colleges of Physicians and of Paediatrics and Child Health. https://www.rcplondon.ac.uk/projects/outputs/every-breath-we-take-lifelong-impact-air-pollution.

[16] http://www.eclaire-fp7.eu/sites/eclaire-fp7.eu/files/eclaire-files/documents/Deliverables/D18_4e.pdf

[17] The term 'chronic mortality' refers to the quantification of effects of long-term exposure to pollutants. 'Acute mortality' refers to effects of short-term exposure.

The risk of damage to ecosystems from deposition of acidity or nutrient nitrogen, or exposure to ozone is commonly described in terms of critical loads for deposited pollutants and critical levels for gaseous pollutants. Analysis has been conducted to convert the change in risk of damage from exceedance to an economic equivalent [16]. However, results are highly uncertain given the very limited literature for valuation that this work was able to base its estimates on. A further constraint was that no account was taken of thresholds for human reliance on ecosystem function.

2.3 Geographic scope

The range over which pollutants can cause harm varies according to their mobility within the environment and the characteristics of their response functions. Mercury is an example of an extremely mobile pollutant, for which analysis has been carried out at the global scale. There is increasing interest in consideration of impacts of fine particles and ozone at a hemispheric scale, though the major part of impacts will occur within the region of emission. The methods followed by the European Commission and European Environment Agency recognise trans-boundary impacts at the scale of the whole European continent due to the provisions of the Convention on Long-Range Transboundary Air Pollution (CLRTAP). The need to consider extended distances around industrial facilities is shown by the finding that to account for 80% of the impacts of air pollution from power plants it is necessary to quantify impacts over a radius of 700km [18]. The methods used to inform analysis for the European Commission and the EEA are carried out at the European scale using information from the European Monitoring and Evaluation Programme (EMEP) [19]. The EMEP domain is shown in Figure 2.

The modelling used for the development of the marginal damage costs presented by the EEA accounts for impacts across the European part of this domain, excluding impacts to Iceland, North Africa, the Middle East, Turkey and the Caucasian countries (Armenia, Azerbaijan and Georgia) and Russia east of the Caucasus Mountains. The results published by the EEA do not therefore account fully for impacts, particularly for countries towards the east and the south of the modelled domain. The EMEP model covers all known sources of air pollution in Europe and is validated against official air quality data from monitoring stations every year.

[18] Preiss P, Roos J, Friedrich R (2013). Estimating Health Risks caused by Emissions of Air Pollutants from Coal Fired Power Plants in Europe - Documentation of Methods and Results. University of Stuttgart. http://www.greenpeace.org/czech/Global/czech/P3/dokumenty/Klima/Estimating_Health_Risks_IER.pdf

[19] http://www.emep.int/

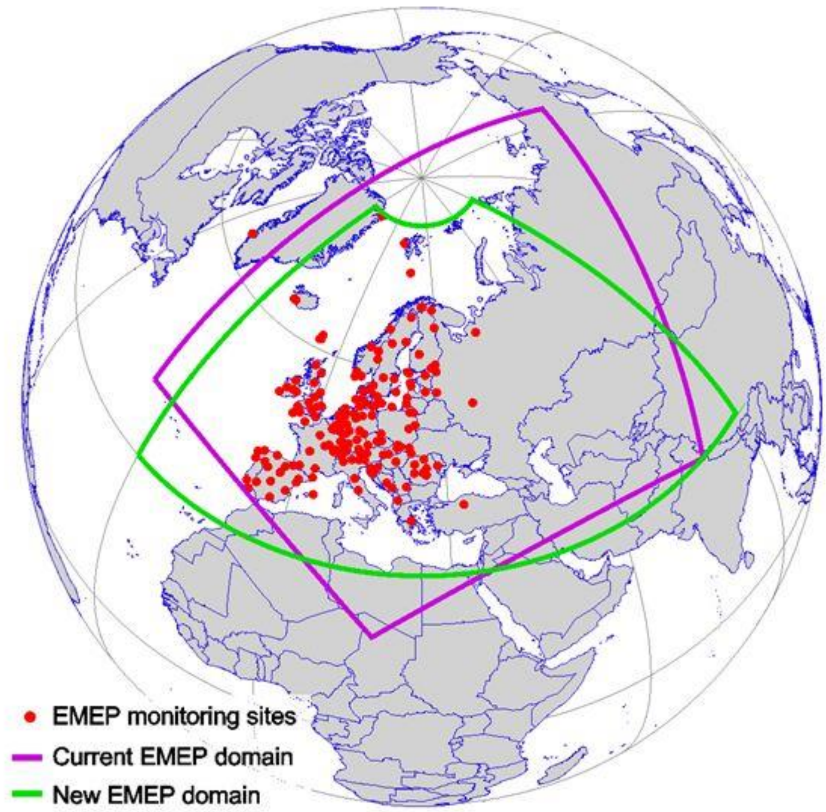

Figure 2. The EMEP domain. Source: http://emep.int/mscw/index_mscw.html.

Quantification of impacts over extended distances will lead to assessment in areas where the statutory ambient air quality limits are not exceeded, as well as areas where these limits are exceeded. However, the distinction between areas where limits are met and those where they are exceeded is largely irrelevant here, as there is no evidence for thresholds for the pollutants of most concern (fine particles and most toxic metals). The most persuasive observations of there being no threshold, taking the example of fine particles, is from analysis in Canada, covering regions of the country with very low pollutant concentrations [20]. Impacts of particle exposure were identified throughout the country. The HRAPIE study of WHO [13] identified 'cut points' for quantification of impacts of NO_2 and O_3. Although identical in effect to the assumption of a threshold, the 'cut-points' simply reflect a lack of data in the epidemiological literature on response to lower concentrations.

The lack of thresholds arises because there will always be some in the population who are sensitive perhaps because they are very young or old, or simply unwell. In some cases, illness will be initiated wholly or partially by air pollution, while with lower levels of air pollution the illness would not have developed. In others it may be caused by other agents. The role of those other risk factors is accounted for in the IPA as the air pollution risk factors are derived from very large epidemiological studies that can exclude confounding risk factors in the statistical analysis.

[20] Villeneuve, P.J., Weichenthal, S.A., Crouse, D., Miller, A.B., To, T., Martin, R.V., van Donkelaar, A., Wall, C., Burnett, R.T. (2015) Long-term Exposure to Fine Particulate Matter Air Pollution and Mortality Among Canadian Women. Epidemiology;26(4):536-45. doi: 10.1097/EDE.0000000000000294.

It is acknowledged that the greatest increase in exposure to air pollution associated with an industrial site may be a few kilometres downwind of the plant. However, whilst it is relevant to consider whether the pollutant levels at this location exceed the ambient air quality limits or not, the concentrations at this point are no guide to wider impacts on health. Given the lack of thresholds the consequence for impacts is not solely related to concentration, but to the product of population and concentration, exposure. Hence a large increase in concentration close to a plant, but in an area with a low population density, may have less health impact than a much smaller change in concentration far away, that affects a large centre of population.

2.4 Time

The rule following from the polluter pays principle is that effects should be quantified over their full time horizon. The time horizon has two components:

- Persistence in the environment. This will be short for highly reactive pollutants (though their decay products may also have impacts), or pollutants that are active only in the air (e.g. fine particles that need to be inhaled to cause damage), but longer for less reactive pollutants such as toxic metals.
- Impacts that are a function of cumulative intake of pollution.

The economic values of impacts quantified over future years (as distinct from the impacts themselves, which should be reported as calculated) require discounting back to the present. The social discount rate adopted by the European Commission is 4%. Higher figures should not be used for the benefits assessment. Account should also be taken of increasing valuations over time reflecting economic growth. This will naturally act in opposition to the discount rate (further information is provided below).

3 Emission levels on which to base quantification

Comparison should be made between emission levels forecast under the application for derogation, and the emission levels that would be achieved using the technology or technologies that would allow compliance with the IED. This will actually be lower than the upper BAT limit, for the following reason: For a new plant complying with its permit, associated emission limits represent a worst case as it is extremely unlikely that a plant operator would either be able to manage the plant to 'just meet' its limits, or that the input materials that will partly determine emission levels will be managed in such a way as to precisely meet the limits. Operators will work with some margin between actual and permitted levels. On this basis, reported or forecast emission levels from comparable facilities may appear to be a more reasonable indication of future emissions than the levels identified in the permit.

However, it is not uncommon for emission limits for some pollutants to be set far in excess (for example, by a factor of 2 or 3) of what is achievable. This situation gives operators flexibility in how limit values are met, meaning that forecast emissions may be exceeded.

The same applies to existing plant. Current measured emissions, or forecast emissions based on experience to date, may be used for the quantification. However, if emission limits are set with too generous a margin, operators may increase emissions beyond anticipated levels. The most robust approach in this situation is to quantify against current emission levels and also against permitted levels. This will highlight areas where an excessive margin between current and potential emissions is requested.

4 Guidance on how to quantify and monetise avoided impacts from air pollution

As noted above, the impact pathway approach illustrated in Figure 1 provides an approach to quantification of the health and other impacts of air pollutant emissions that is universally accepted as the correct method. When applied in full for a specific site it is able to account for the characteristics of the plant, such as stack height and flue gas volume and temperature that will influence dispersion, local topography and local climate. the full application of the IPA is not straightforward, requiring knowledge of a number of models and disciplines.

For these reasons, most analysts will find it easier to apply the marginal damage costs per tonne emission of each pollutant, as used by the EEA. This section describes how those values were derived. The information provided also defines the full IPA.

4.1 Pollutant dispersion

With characterisation of emissions being described in Section 0, we start this section by considering dispersion modelling. Many dispersion models are available [21], though only a few are suitable for the work discussed here. Much of the modelling of emissions from industrial sites has considered the spread of pollutants over a short range, largely to identify:

- The maximum contribution to ground level concentrations
- Any areas where emissions from a plant may be a major factor leading to exceedance of ambient air quality limits

Given these objectives, many models operate over short ranges and often fail to account for the reactivity of pollutants. Some models allow users to perform calculations over a range of meteorological conditions, permitting assessment of 'worst case' scenarios.

As noted above, more sophisticated models are required to quantify something close to the full impacts of pollutant emissions, taking account of:

- Impacts over very long ranges (going to several hundred km)
- Pollutant reactivity
- Meteorology
- Extended timescales, with results needing to be averaged over a year to link with the response functions used

Meteorology can be factored into the modelling by using data from a number of years, covering both 'typical' years and extremes. This greatly increases the time required to run the model.

A number of models have been developed for this work in Europe, including:

- CHIMERE
- EMEP (the model used in European policy assessments and for the damage costs provided by the EEA)
- EURAD-IM
- LOTOS-EUROS
- MATCH
- MOCAGE

[21] https://en.wikipedia.org/wiki/List_of_atmospheric_dispersion_models

- SILAM

The ENSEMBLE system seeks to integrate results from across this set of models and provides documentation on each of them [22]. The availability of these models for analysis of specific emission sources may be very limited.

A consequence of using models operating at the continental scale is that results are calculated to a rather coarse grid, typically of the order 25x25km. Whilst these models are thus not well suited to identifying locations where air quality limits may be exceeded, or where ground level concentrations will be highest, problems become much less important when dealing with exposure of the population as a whole for the type of plant likely to be of most relevance here, given that pollutants will be released from tall stacks, reducing the importance of local topography and very localised climate factors, and that much of the impacts of interest are linked to secondary pollutants that will take some time to form in the atmosphere. Both factors will work to smooth the concentration of pollutants over the modelled area.

The EEA's marginal damage costs were based on dispersion calculations made by the EMEP model. Country to country source-receptor matrices were developed from a large number of model runs, for each country:

1 baseline run at current emissions x 5 meteorological years
+
4 runs to account for each pollutant (NH_3, NOx + primary $PM_{2.5}$, SO_2, VOCs) x 5 meteorological years

Primary $PM_{2.5}$ was considered unreactive and hence modelled with NOx, to reduce the amount of computer time needed. For the pollutant-specific runs, emissions were reduced by 15% relative to the baseline from the country for which analysis was being conducted. Emissions from all other countries were held constant.

4.2 Exposure of sensitive receptors

Once data are available from dispersion models it is a routine matter to combine results with information on the distribution of the human population on a similar grid scale. Many of the health-response functions require population data by age group:

Information on the production of crops and forests is available at the national level through Eurostat and FAO, though more detailed datasets are necessary for subnational modelling. For ecosystems, information on critical loads exceedance is available through sources linked to the UN/ECE Convention on Long Range Transboundary Air Pollution. The ecosystems data is well linked to some of the models listed above (e.g. EMEP) as ecosystem damage was one of the main drivers for the development of transboundary air pollution legislation in Europe.

There is a lack of up to date information on materials, with the last inventories having been compiled in the 1990s. It is generally considered that these are still a reasonable approximation, though this assumption has not been tested with new data collection. The issue is of little importance given that materials damage provides only a small part of total quantified damage, accepting that impacts on cultural heritage remain unquantified.

[22] https://atmosphere.copernicus.eu/documentation-regional-systems

4.3 Modelling impacts

Guidance on the quantification of health impacts of air pollution, drawing on the conclusions of the WHO-led HRAPIE study [13] and covering all of the endpoints identified there for quantification, is available in a report by Holland (2014) produced for the European Commission in the context of the Clean Air Policy Package of 2013 [23]. The report, which was used for the quantification of the EEA damage costs, provides information on:

- Response functions
- Part of the population associated with each response function
- Incidence data for each health effect
- Further necessary data, such as days of sick leave, by country.

A further report by Holland (2014) provides information on the application of these functions for the scenarios considered under the Clean Air Policy Package, and includes the unit values applied for each effect [24].

Table 1 lists the impacts recommended for quantification by the WHO HRAPIE study. Analysis for the European Commission has considered all of these effects since the publication of the HRAPIE study, with the exception of impacts associated with exposure to NO_2. Whilst there is sufficient evidence to conclude that there are impacts, in particular on mortality, beyond those quantified for exposure to the ambient pollution mix, the precise role of NO_2 is considered unclear, and appropriate scales for modelling are still under debate, together with the approach for avoiding double counting when quantifying effects of exposure to NO_2 and fine particles.

Quantification of effects on crops (reduced productivity), forests and ecosystems is described in reports of Work Package 18 of the ECLAIRE study [25]. ECLAIRE succeeded in quantifying lost production of all arable crops in the EU from chronic ozone exposures. Impacts on grasslands, leading to impacts on production of livestock, milk, etc. were outside of the analysis, but thought likely to be of lower importance. For forests, ECLAIRE quantified lost production of timber, and reduce carbon sequestration. Impacts on forests are yet to be included in the EEA estimates.

The approach for quantification of impacts on materials is described by Rabl et al (2014) [26], in a book that discusses methods and result of air pollution benefit assessment in extensive detail. A simplistic approach was taken to inclusion of material damage costs in the EEA estimates, using figures derived per tonne of SO_2 only. Effects of NOx emissions, either directly or through ozone formation are small. Effects of soiling from the deposition of particles are more significant, but again small compared to human health impacts.

[23] http://ec.europa.eu/environment/air/pdf/CBA%20HRAPIE%20implement.pdf

[24] http://ec.europa.eu/environment/air/pdf/TSAP%20CBA.pdf

[25] The ECLAIRE methods for benefits assessment are described in a series of 4 reports: http://www.eclaire-fp7.eu/sites/eclaire-fp7.eu/files/eclaire-files/documents/Deliverables/D18_1.pdf, http://www.eclaire-fp7.eu/sites/eclaire-fp7.eu/files/eclaire-files/documents/Deliverables/D18_2.pdf, http://www.eclaire-fp7.eu/sites/eclaire-fp7.eu/files/eclaire-files/documents/Deliverables/D18_3.pdf , http://www.eclaire-fp7.eu/sites/eclaire-fp7.eu/files/eclaire-files/documents/Deliverables/D18_4e.pdf

[26] Rabl, A., Spadaro, J. and Holland, M. (2014) How Much Is Clean Air Worth? Cambridge University Press.

Table 1. List of health impacts – HRAPIE recommendations.

Impact / population group	Rating	Population	Exposure metric
All cause mortality from chronic exposure	B	Over 30 years	O_3, SOMO35, summer months
All cause mortality from acute exposure	A*/A	All ages	O_3, SOMO35 (A*), SOMO10 (A)
Cardiac and respiratory mortality from acute exposure	A	All ages	O_3, SOMO35 (A*), SOMO10 (A)
Respiratory Hospital Admissions	A*/A	Over 65 years	O_3, SOMO35 (A*), SOMO10 (A)
Cardiovascular hospital admissions	A*/A	Over 65 years	O_3, SOMO35 (A*), SOMO10 (A)
Minor Restricted Activity Days (MRADs)	B*/B	All ages	O_3, SOMO35 (B*), SOMO10 (B)
All cause mortality from chronic exposure as life years lost or premature deaths	A*	Over 30 years	$PM_{2.5}$, annual average
Cause-specific mortality from chronic exposure	A	Over 30 years	$PM_{2.5}$, annual average
Infant Mortality	B*	1 month to 1 year	$PM_{2.5}$, annual average
Chronic bronchitis in adults	B*	Over 27 years	$PM_{2.5}$, annual average
Bronchitis in children	B*	6 – 12 years	$PM_{2.5}$, annual average
All cause mortality from acute exposure	A	All ages	$PM_{2.5}$, annual average
Respiratory Hospital Admissions	A*	All ages	$PM_{2.5}$, annual average
Cardiovascular Hospital Admissions	A*	All ages	$PM_{2.5}$, annual average
Restricted Activity Days (RADs)	B*	All	$PM_{2.5}$, annual average
Including lost working days	B*	15 to 64 years	$PM_{2.5}$, annual average
Asthma symptoms in asthmatic children	B*	5 to 19 years	$PM_{2.5}$, annual average
All cause mortality from chronic exposure	B*	Over 30 years	NO_2 annual mean >20ug.m^{-3}
All cause mortality from acute exposure	A*	All ages	NO_2 annual mean
Bronchitis in children	B*	5 – 14 years	NO_2 annual mean
Respiratory hospital admissions	A*	All ages	NO_2 annual mean

Notes: Rating column: A = effects that can be quantified with greatest confidence, B = effects that can be quantified but with less confidence, * = effects that should be included in CBA.

4.4 Valuation of impacts

Health values used for the EEA calculations were taken from the study on the EC's proposals for the Thematic Strategy on Air Pollution, and are given in Table 2 [24].

Crops were valued in ECLAIRE using prices on the world market, avoiding to some extent distortions arising from the Common Agriculture Policy of the EU [27]. Changes in forest production from ozone exposure were valued using market data from Eurostat [28]. The carbon price for addressing changes in carbon sequestration by forests was set at €9.5/t CO_{2eq} and €38.1/t CO_{2eq} . However, these data are yet to be included in the EEA damage costs.

The range of values has been used to value changes in carbon sequestration linked to ozone exposure of forests, following the figures adopted by the EEA [1]:

- A lower value of €9.5 per tonne CO_2 (2005 price), based upon a value of €10 per tonne CO_2 in 2010 prices reflecting the modelled ETS (Emissions Trading Scheme) price in 2020 based on a reference scenario based on implementation of current legislation;

[27] http://faostat3.fao.org/download/Q/QV/E
[28] http://appsso.eurostat.ec.europa.eu/nui/show.do.

- A higher value of €38.1 per tonne CO_2 (2005 price), based upon a value of €40 per tonne CO_2 in 2010 prices) reflecting the projected carbon price in 2030 in a central scenario of 40% domestic GHG emission reduction by 2030 compared to 1990 [29].

Table 2. Values used for the health impact assessment (price year 2005)

Impact / population group	Unit cost	Unit
Ozone effects		
Mortality from chronic exposure as: Life years lost, or Premature deaths	57,700 / 133,000 1.09 / 2.22 million	€/life year lost (VOLY) €/death (VSL)
Mortality from acute exposure	57,700 / 138,700	€/life year lost (VOLY)
Respiratory Hospital Admissions	2,220	€/hospital admission
Cardiovascular Hospital Admissions	2,220	€/hospital admission
Minor Restricted Activity Days (MRADs)	42	€/day
$PM_{2.5}$ effects		
Mortality from chronic exposure as: Life years lost, or Premature deaths (all-cause and cause-specific mortality)	57,700 / 133,000 1.09 / 2.22 million	€/life year lost (VOLY) €/death (VSL)
Mortality from acute exposure	57,700 / 138,700	€/life year lost (VOLY)
Infant Mortality	1.6 to 3.3 million	€/case
Chronic Bronchitis in adults	53,600	€/new case of chronic bronchitis
Bronchitis in children	588	€/case
Respiratory Hospital Admissions	2,220	€/hospital admission
Cardiac Hospital Admissions	2,220	€/hospital admission
Restricted Activity Days (RADs)	92	€/day
Work loss days	130	€/day
Asthma symptoms, asthmatic children	42	€/day
NO_2 effects (though not quantified in this report)		
Mortality from chronic exposure as: Life years lost, or Premature deaths	57,700 / 133,000 1.09 / 2.22 million	€/life year lost (VOLY) €/death (VSL)
Mortality from acute exposure	57,700 / 138,700	€/life year lost (VOLY)
Bronchitis in children	588	€/case
Respiratory Hospital Admissions	2,220	€/hospital admission

Repair costs for building materials were taken from standard sources used for costing work by architects and builders.

The values selected for the analysis need to be adjusted to represent values in the same year as used for the analysis of control costs. The method for doing this is described in Section 5.3.

[29] http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52014SC0015&from=EN

5 Evaluation of marginal external damage costs of air pollution provided by the European Environment Agency

5.1 Overview of the EEA estimates

Key features of the analysis carried out for the EEA to derive marginal damage costs per tonne of pollutant are as follows:

1. Estimates were derived using the impact pathway approach.
2. Pollutant dispersion and chemistry for NH_3, NOx, $PM_{2.5}$, SO_2 and VOCs was described using the transfer matrices developed using the EMEP model, which underpins air pollution policy work for the European Commission and UN/ECE under the Convention on Long Range Transboundary Air Pollution.
3. Pollutant dispersion for toxic metals and trace organics was performed using the Uniform World Methodology of the RiskPoll model. The model accounts for human exposure to these pollutants from inhalation and dietary intake.
4. Health impacts were quantified using the response functions recommended in the WHO HRAPIE study.
5. Results from quantification of $PM_{2.5}$ impacts were extrapolated to PM_{10} using a factor of 0.65. This assumes that health impacts are caused by the $PM_{2.5}$ fraction only, and that 65% of PM_{10} is $PM_{2.5}$. The results for the two fractions are not additive. In the event that analysts have an alternative estimate of the fraction of PM_{10} that is $PM_{2.5}$ it is suggested that this factor is applied instead of 65%.
6. Health valuation was carried out using the same data as used in analysis of air quality policies for the European Commission.
7. Crop damage was included for emissions of NOx and VOCs, via ozone formation.
8. Materials damage was included for emissions of SO_2.
9. No account was taken of damage to forests and natural ecosystems or effects on visibility [30].
10. Health impacts of NO_2 were not included. Results for NOx and PM can therefore be added without applying the 30% correction suggested by HRAPIE.

The EU marginal damage figures were supplemented by the use of additional factors to account for the height of release from industrial sources, derived from the Eurodelta II study, which provided a comparison of a number of European-scale dispersion models. An improved approach using information from the SHERPA tool is currently under evaluation for this work [31].

EU average results are shown in Table 3, including adjustment with average figures to account for release of pollutants from tall stacks.

[30] Impacts on visibility have long been considered a major concern for impact assessments in the USA, but have raised little interest in Europe.

[31] http://aqm.jrc.ec.europa.eu/sherpa.aspx

Table 3. EU average marginal damage costs per tonne of pollutant emitted (EUR per tonne, 2005 prices, with Eurodelta adjustment)

	NMVOCs	NOx	NH_3	SO_2	PM_{10}	$PM_{2.5}$
EU-28 average, lower range (median VOLY)	1,369	4,664	9,259	11,170	17,858	27,501
EU-28 average, upper range (mean VSL)	3,574	12,586	27,238	32,754	52,235	80,443
Eurodelta II correction factors, average and range (specific correction factors available for FR, DE, ES, UK)		0.78 ± 0.13		0.87 ± 0.14	0.5 ± 0.14	0.5 ± 0.14
After correction factor (median VOLY)						
After correction factor (mean VSL)		9,817 (±1,636)		28,496 (±4,586)	26,118 (±7,313)	40,222 (±11,262)

Notes: Country-specific values can be extracted from EEA 2014.
Results for $PM_{2.5}$ and PM_{10} are not additive
Analysis of specific facilities should use the country specific numbers provided in the EEA report, with adjustments as described below, rather than European averages.

5.2 Completeness

5.2.1 Pollutants covered

The EEA dataset provides results for:

- Major pollutants:
  - NH3, NOx, PM2.5, PM10, SO2, VOCs
  - Note: For NOx, health impacts are included only from exposure to fine particles derived from NOx emissions, and do not include direct effects of NO_2.
  - Note: For 'VOCs' (collectively), only effects from exposure to ozone are included.
- Organics:
  - Benzene, formaldehyde, dioxins and furans, 1,3 butadiene, diesel exhaust, PAHs
  - Note: Results specific to these pollutants do not include impacts via the formation of ozone (where appropriate).
- Toxic metals:
  - Arsenic, cadmium, chromium, lead, mercury, nickel
- Secondary pollutants
  - Secondary inorganic and organic aerosols, ozone

Further to the EEA values for metals, a series of papers by Nedellec and Rabl [32] have been published recently (see Table 4 below). These include a larger number of effects than those considered in the EEA report and thus provide higher values. These new results should be taken seriously as they are more recent and have been published in the peer reviewed literature.

[32] Nedellec V. and Rabl A. (2016) Costs of Health Damage from Atmospheric Emissions of Toxic Metals: Part 2-Analysis for Mercury and Lead. .Risk Anal. 2016 Nov;36(11):2096-2104. doi: 10.1111/risa.12598. Epub 2016 Mar 14.
Nedellec V. and, Rabl A. (2016) Costs of Health Damage from Atmospheric Emissions of Toxic Metals: Part 1-Methods and Results. Risk Anal. 2016 Nov;36(11):2081-2095. doi: 10.1111/risa.12599. Epub 2016 Mar 10.

However, they also address impacts for which the current literature is limited. It is recommended here that the range from the EEA figures to Rabl et al is applied when considering derogations that are specific to metals, or where a facility for which derogation is applied is a major source of metals that could be avoided through (e.g.) better capture of particles.

5.2.2 Impacts addressed

Major pollutants

The marginal damage costs account for all of the health impacts identified in the WHO HRAPIE study, except for those associated with exposure to NO_2 specifically (secondary pollutants associated with NOx emissions, nitrate aerosols and ozone, are considered, but not NO_2).

Crop damage and materials damage are addressed for the pollutants of most concern for them (NOx and VOCs, and SO_2, respectively).

VOCs, as a class of pollutants, are only assessed in terms of their effects on ozone levels and their contribution to secondary organic aerosols. Individual VOCs may have a range of additional impacts on health and the environment.

Damage to forests, natural ecosystems and effects on visibility are not considered. The omission of these impact categories seems unlikely to make a substantial difference to most assessments, with damage costs for forests and natural ecosystems being equivalent to around only 5% of health impacts when a conservative (low) valuation of mortality is adopted [25]. Effects on visibility are also unlikely to make much difference to the damage costs, given that it is not an impact that has attracted any concern in Europe.

Trace pollutants (organics and metals)

Effects for these pollutants are focused in the EEA analysis on carcinogenicity and for lead and mercury, on IQ loss. Nedellec and Rabl bring in additional effects for the metals, including a broader assessment of mortality, leading to a substantial increase in damage costs.

Table 4. Estimates of external costs per kg emission for various metals from Nedellec and Rabl (2016), with damage costs from the EEA report shown for comparison in the final row of the table.

	Arsenic	Cadmium	Mercury	Lead
Non cancer mortality	2,120	136,752	20,833	24,858
Cancer deaths	720			
Non fatal cancers	41	421		
Chronic bronchitis	194			
IQ loss	645		2,104	4,435
Infant deaths	31			
Diabetes	1,962			
Fractures		1,796		
Anaemia				50
Total, €/kg	**5,700 – 7,100[1]**	**140,000**	**23,000 – 52,000[1]**	**29,000**
Total from EEA report	**30 - 530**	**5.2 – 47**	**80 – 4,290**	**90 – 1,480**

Some of the organics (e.g. benzene) make a contribution to ozone levels, but this is not considered in the EEA's analysis of trace organic pollutants.

5.2.3 Valuation

The largest uncertainty affecting the damage costs published by the EEA concerns the valuation of mortality, which accounts for between 70% and 95% of total damage. The median VOLY (value of a life year) to mean VSL (value of statistical life) range adopted for the EEA estimates follows the valuations adopted under the European Commission's CAFE programme of 2005.

The respective unit values are applied to different indicators of mortality burden, with VOLY applied to an estimate of the aggregate loss of life expectancy (life years) and VSL applied to an estimate of the number of deaths associated with air pollution. Interpretation of 'air pollution deaths' is not straightforward. Those affected are likely to also be affected by other burdens on health, for example from smoking, lack of exercise, poor diet and so on. The UK's Committee on the Medical Effects of Air Pollutants (COMEAP) preferred to use the term 'equivalent attributable deaths' to 'deaths', noting that the total number of people whose death is in some way linked to air pollution was likely to be substantially higher than the numbers indicate, but that those deaths could not be solely attributed to air pollution [33]. The distinction between 'deaths' and 'equivalent attributable deaths' is important, providing some rationale for returning to use of the VSL in preference to the VOLY.

Peer review of the CAFE benefit assessment methods concluded that: "*For the CAFE CBA methodology, the independent external peer reviewers and several stakeholders suggested that both the VSL and the VOLY approaches be used, to show transparently the variation in results arising from use of these two approaches*."

Different economists prefer different metrics for the expression of willingness to pay to avoid loss of life expectancy. The European Commission and European Environment Agency in their cost or cost-benefit assessments use both metrics complementarily as an expression of sensitivity of cost estimates. The OECD strongly prefers the VSL approach [34] and provided an updated estimate of the VSL in 2012 of €3 million in the EU28. Adjusting for inflation means that this figure is 27% higher than the upper end of the VSL used in the EEA analysis. The OECD work was funded by the European Commission and it is notable that VSL estimates of a similar magnitude (and higher) have been used in recent analysis by the European Chemicals Agency (ECHA) under the REACH Regulation addressing the risks of hazardous chemicals, suggesting that what is regarded as an upper bound for air quality impact assessment is elsewhere regarded as conservative. This also implies that the Commission's estimate of the VOLY is also conservative.

Further questions concern the use of median rather than mean estimates of willingness to pay, particularly given the use of screening techniques to eliminate very high estimates (outliers) that will unduly influence the mean. Use of the median naturally gives lower values, and in doing so ignores the views of some members of society.

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[33] https://www.gov.uk/government/publications/comeap-long-term-exposure-to-air-pollution-effect-on-mortality

[34] http://www.oecd.org/environment/mortalityriskvaluationinenvironmenthealthandtransportpolicies.htm

On the other hand, the one notable publication of an estimate of the VOLY [35] in recent years gives a lower figure (€40,000) than the 'median VOLY' accepted by the Commission for air quality impact assessment. However, the VSL derived from the same study is low compared to the wider literature, based on the result of the OECD meta-analysis, and so it appears also that the €40,000 will be an underestimate. It is thus not recommended for adoption in analysis of derogations to the IED.

5.2.4 Transboundary impacts

As noted in Section 2.3, the modelling used for the development of the marginal damage costs presented by the EEA accounts for impacts across the European part of this domain, excluding impacts to Iceland, North Africa, the Middle East, Turkey and the Caucasian countries (Armenia, Azerbaijan and Georgia) and Russia east of the Caucasus Mountains. The results published by the EEA do not therefore account fully for impacts, particularly for countries towards the east and the south of the modelled domain.

5.3 Adjustment of published values

There are several adjustment that can be made to the damage cost data, covering the following factors:

- Price year
- Discounting and other temporal adjustments
- National vs EU-wide figures
- Overlap between $PM_{2.5}$, PM_{10}, and NO_2
- Sector-specific characteristics

5.3.1 Price year

Inflation is typically accounted for in Europe using the Harmonised Index of Consumer Prices. Data from 1997 to 2017 are presented in Table 5. The table uses 2005 as the base year (index = 100) as this is the price year used by the EEA.

5.3.2 Discounting and other temporal adjustments

Future damage should be discounted at a rate of 4%, in line with European Commission practice.

Although incomes per capita across the EU have been static in recent years, adjustment should be made for future income growth, recognising that this will increase willingness to pay for health protection. Using estimates of GDP/capita for each country out to 2050, weighted average growth rates for the EU based on OECD data are shown in Table 6. It is suggested that these growth rates are subtracted from the 4% discount rate to give the adjusted discount rate in the bottom row of the table.

[35] http://www.sciencedirect.com/science/article/pii/S1470160X10002116

Table 5. Change in HICP, 1997 to 2017

	HICP inflation	Index, base year 2005
1997	1.71%	85.6
1998	1.23%	86.6
1999	1.17%	87.6
2000	2.18%	89.5
2001	2.41%	91.7
2002	2.27%	93.8
2003	2.12%	95.8
2004	2.18%	97.8
2005	2.20%	100.0
2006	2.21%	102.2
2007	2.16%	104.4
2008	3.35%	107.9
2009	0.32%	108.3
2010	1.61%	110.0
2011	2.72%	113.0
2012	2.50%	115.8
2013	1.35%	117.4
2014	0.43%	117.9
2015	0.03%	117.9
2016	0.24%	118.2
2017	1.56%	120.1

Data source: http://www.inflation.eu/inflation-rates/europe/historic-inflation/hicp-inflation-europe.aspx

To convert from 2005 to 2017 prices, multiply the 2005 data by a factor of 120.1/100 (=1.201). Similarly, to convert from 1999 prices to 2014, apply a factor of 117.9/87.6 (=1.345).

Table 6. Averaged GDP uplift, discount rate and adjusted discount rate 2010 to 2050 for EU member states.

	2010-2020	2020-2030	2030-2040	2040-2050
GDP uplift	1.56%	2.06%	1.55%	1.32%
Discount rate	4%	4%	4%	4%
Adjusted discount rate	2.44%	1.94%	2.45%	2.68%

For simplicity, an average adjusted discount rate of 2.4% could be used for the whole period.

5.3.3 National vs EU-wide figures

Given that air pollution impacts are transboundary, conversion of EU average values to local values (e.g. applying ratios of GDP/capita adjusted for purchasing power parity) will provide a misleading result, unless similar adjustments are applied to other countries in a way that reflects their own PPP adjusted GDP/capita relative to the EU's. This would require full application of the IPA in order to determine how the damage from the country of origin is distributed around Europe. Adoption of EU-wide averages is a simpler option.

5.3.4 Overlap between $PM_{2.5}$, PM_{10}, and NO_2

The damage costs provided by the EEA for $PM_{2.5}$ and PM_{10} are not additive, but provided to facilitate easy calculation of damage for different measures of particulate matter. The PM_{10} estimates of damage per tonne were derived simply by multiplying the $PM_{2.5}$ result by a factor 0.65, assuming that roughly 65% of PM_{10} from industrial sources is in the $PM_{2.5}$ fraction which is considered the most harmful part as it can penetrate deep into the lung. Given the way that the epidemiology studies are conducted, quantifying against $PM_{2.5}$ implicitly accounts for any impact of other particle sizes assuming that they are correlated with $PM_{2.5}$ mass. Accordingly:

- If PM data are provided as $PM_{2.5}$ use the damage/tonne estimate for $PM_{2.5}$.
- If they are provided as PM_{10} use the damage/tonne estimate for PM_{10}.
- If both $PM_{2.5}$ and PM_{10} data are provided use either estimate but do not add them together.

The situation with NO_2 is more complex, as some part of the damage quantified against NO_2 exposure will be additional to effects associated with fine particles. Evidence suggests that NO_2 exposure generates the second highest impact on health after PM. However, at the present time, the EEA damage costs take no account of impacts specific to NO_2, reflecting difficulty in modelling NO_2 exposure and determining the extent of overlap between the PM and NO_2 functions. This leads to an underestimation of impacts attributable to NOx emissions (though the damage/tonne functions for NOx include account of its contribution to particle concentrations via nitrate aerosol formation). Analysis of compliance with air quality limit values should still account for NO_2 where derogation is sought for NOx.

5.3.5 Sector-specific characteristics

The damage costs per tonne presented by the EEA are national averages, accounting for all sources. Industrial sources tend to be less closely linked to population than some others (e.g. traffic) and hence it is anticipated that they will have lower damage costs per unit emission. The EEA applies damage costs from the Eurodelta II study (see Table 3). The Eurodelta data are limited. A more extensive dataset may be available via the SHERPA study [31], though this has not yet been evaluated here.

6 Guidance on evaluating the appropriateness of cost estimates for retrofit costs provided by operators or authorities

Guidelines for quantifying the costs of abatement measures have been published by the EEA[36]. Although these guidelines are nearly 20 years old, the approach that they define has not changed in the intervening years.

Cost data can be checked (to some degree) against estimates provided in the various BREF notes published by the European IPPC Bureau in Sevilla [37]. Costs will be most reliable for new plant. Costs for retrofitting equipment will typically be higher because of the need to develop new systems within an existing plant layout. In the event that retrofit costs seem significantly higher than published estimates, data should be challenged to understand why the difference in cost exists. Other sources of cost data include:

- Information used in the GAINS model [38]
- A report from the Task Force on Techno Economic Issues reporting to the Convention on Long Range Transboundary Air Pollution [39].

Analysis should be fully transparent with respect to the parameters used in analysis (discount rate, etc.) and the source of abatement cost data. An argument of commercial confidentiality may be made, but can be countered from the perspective that the applicant is seeking to trade their interests against human and environmental health: it is not unreasonable that those affected by the decision (or their representatives) are given full access to information.

Assumptions on plant lifetime are a major determinant of the outcome of cost-benefit analysis of abatement technologies. The shorter the period considered, the less the likelihood that benefits of action will exceed costs. There are no firm rules for evaluating the period adopted by an applicant for derogation: assessment will need to be done on a case by case basis. It may be useful to consider the period required for benefits to exceed costs, if this is not provided, and for alternative scenarios of benefits.

The cost of abatement technologies should decline over time for several reasons, including:

- Equipment becomes more efficient as experience in its use grows, and as further R&D identifies ways of improving performance
- Research and development costs are paid off
- Competitors enter the market for providing and installing equipment

It may be expected that costs will fall quite quickly once a technology enters the market, but more slowly thereafter. Given that the techniques for LCP controls have been available for some time (use of ESPs and bag filters for particulates, flue gas desulphurisation for SO_2, various de-NOx options), it may be anticipated that the potential for future cost reductions will be small.

[36] https://www.eea.europa.eu/publications/TEC27/download
[37] http://eippcb.jrc.ec.europa.eu/reference/ , with the detailed draft BREF for LCP available at http://eippcb.jrc.ec.europa.eu/reference/BREF/LCP_FinalDraft_06_2016.pdf.
[38] http://www.iiasa.ac.at/web/home/research/researchPrograms/air/GAINS.html
[39] http://www.unece.org/fileadmin/DAM/env/documents/2012/EB/ECE.EB.AIR.117_AV.pdf .

7 Defining disproportionality of retrofit costs in comparison to benefits from avoided air pollution.

There are two scenario based approaches for quantifying the benefits of action:

1. When applying the full impact pathway analysis, impacts would be quantified under scenarios with and without the abatement technology under investigation in place. Subtracting results for the 'with technology' scenario from the 'without technology' scenario gives the benefits associated with the applied technology.
2. When using the marginal damage cost (€/tonne emission) approach, the analysis is simpler: emission reductions that would arise if BAT were implemented (considering the actual emissions that would be achieved by specified technologies, rather than just the upper BAT limit) are multiplied by the unit damage costs for each pollutant.

Disproportionality of retrofit costs is then a simple question of whether the costs of retrofit exceed associated benefits. A number of issues should be noted, all of which have been discussed already:

- Irrespective of the time period adopted for assessment of the costs of added abatement technologies (which may be a function of internal accounting processes within a company), benefits should be quantified for the full future lifetime for the equipment adopted. Consideration should be given to establish the validity of estimates of remaining plant life – the shorter the period, the less the likelihood that benefits of action will exceed costs.
- The discount rates used for benefits should not exceed the 4% adopted by the European Commission.
- Economic growth will increase willingness to pay, providing an offset that works against discounting, with GDP growing at roughly 2% annually in the EU out to 2030. Factoring this into the equation effectively halves the 4% discount rate.
- There are a number of biases in the benefits assessment, discussed above (omission of some important impacts, lack of account of future growth of an ageing population, conservatism in unit damage values, etc.), that bias to underestimation.
- Economic approaches are well developed for health impacts, but not for ecological damage. In particular, the economics does not currently account for irreversible effects in a satisfactory way.

These and other biases should be taken into account in considering whether or not society is best served by allowing a derogation or refusing it. Recognising the omission of several types of impact, and conservatism in the valuations adopted by the EEA it is insufficient to accept that any excess of abatement cost over benefit is sufficient to justify a derogation. In line with the precautionary principle and the polluter pays principle, a simple rule of thumb can be specified, such that a derogation could be granted if:

$$Abatement\ cost > \frac{benefits\ of\ abatement}{0.7}$$

This should be applied for the differing EEA positions on mortality valuation (median VOLY to mean VSL). The question then arises of which position is to be preferred, VOLY or VSL. It is important to bear in mind the discussion above on mortality estimates (Section 5.2.3), regarding how the values adopted by the European Commission are low compared to those recommended elsewhere, with particular emphasis placed on the meta-analysis carried out for

OECD given that it draws on a very extensive literature (the Commission's analysis, in comparison, draws on only 2 studies).

A further perspective on this question can be gained from the wording of the IED regarding the cost-benefit analysis:
"the achievement of emission levels associated with the best available techniques as described in BAT conclusions ***would lead to disproportionately higher costs compared to the environmental benefits****"*
This indicates that the applicant needs to demonstrate that costs would be (not may be) disproportionately higher than benefits. From this perspective, the higher damage cost figure, based on mean VSL is to be preferred for the comparison.