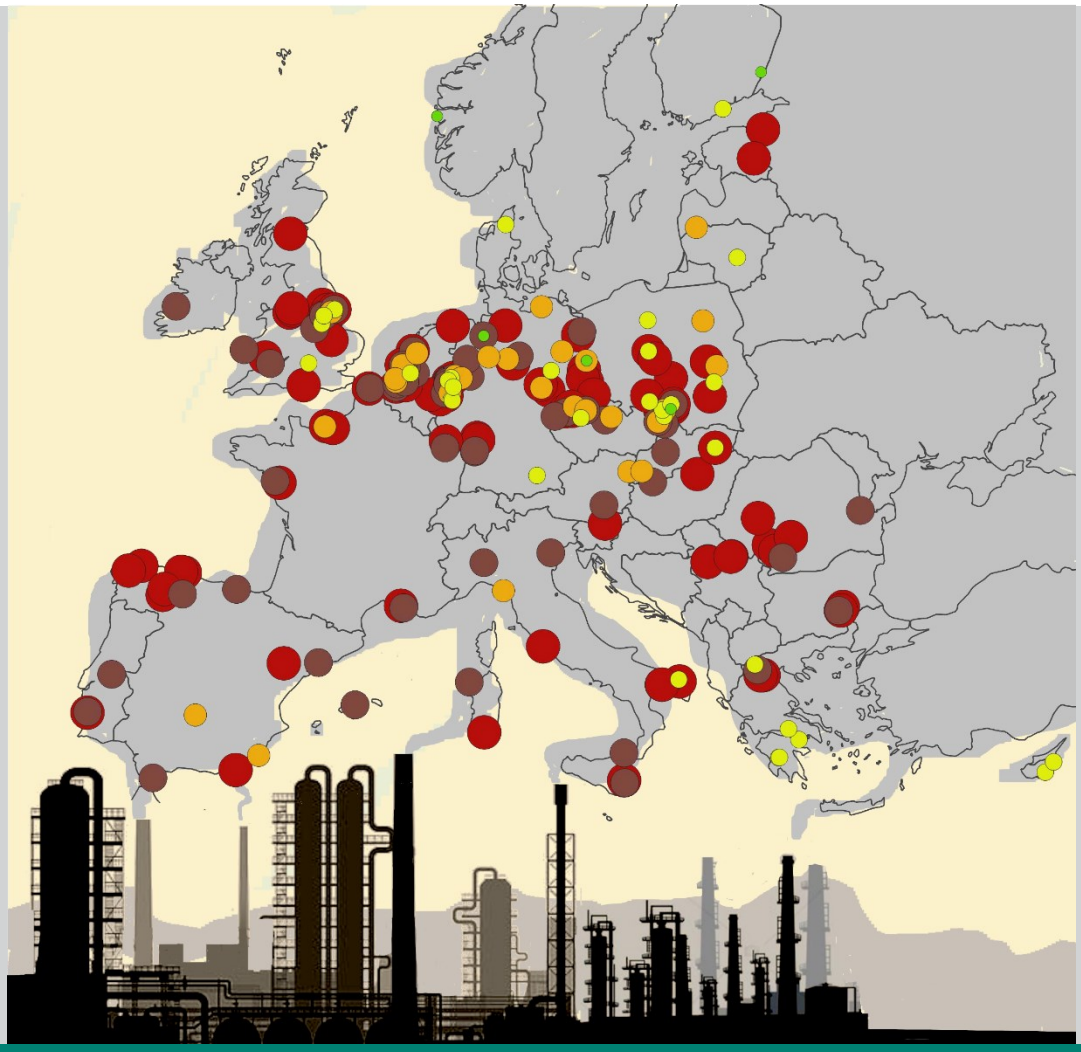
Eionet Report - ETC/ATNI 2019/18

Development of a refined methodology for the EEA externalities assessment

February 2021



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The withdrawal of the United Kingdom from the European Union did not affect the production of the report.
Data reported by the United Kingdom are included in all analyses and assessments contained herein, unless otherwise indicated.

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Executive summary

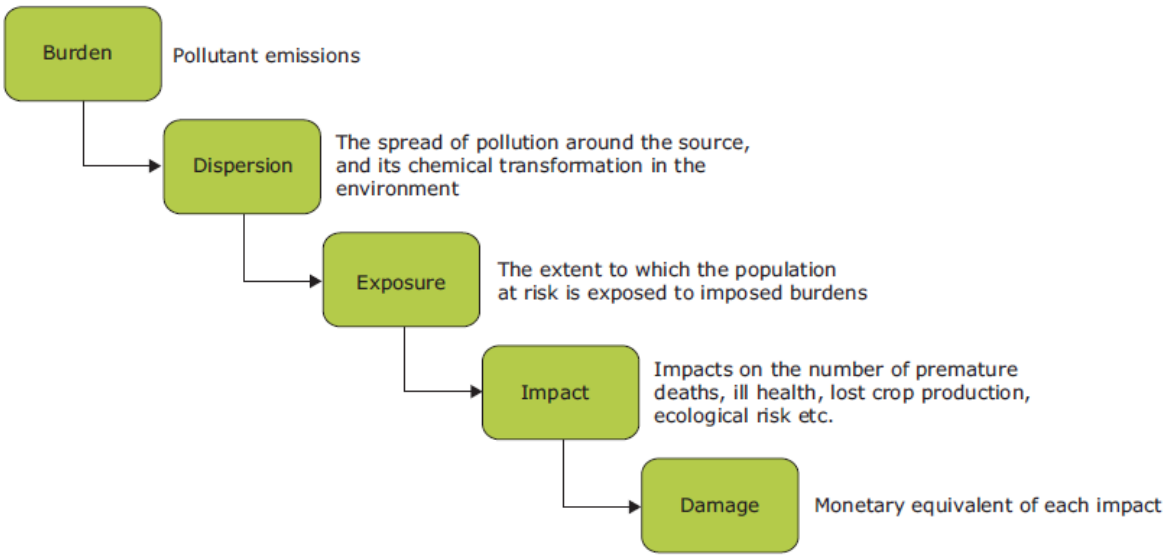
The EEA has published two reports assessing the costs of air pollution from EU industrial facilities (EEA, 2011 & 2014). The assessments involved two stages: a calculation of damage costs per tonne of pollutant emitted and per country, and an assessment of externalities of industrial facilities by multiplying their emissions with the damage costs per tonne of pollutant.

These reports were carried out based on best practice at the time, with the 2014 report presenting an updated assessment of the 2011 report. The EEA is planning a further update of this assessment in 2020. In advance of initiating the assessment the EEA identified a need to critically review the methods used in the 2014 report and identify areas where improvements could be made to further strengthen the outputs from the assessment.

The EEA invited its European Topic Centre on air pollution, transport, noise, and industrial pollution (ETC/ATNI) to carry out this review in 2019. The team involved INERIS and Aether with EMRC as an external consultant, as well as NILU as internal reviewer.

The review covered the whole impact pathway assessment (IPA) route as illustrated in Figure 1, from emissions through exposure to the quantification of health and environmental impacts and their valuation, as well as the consideration of uncertainties. Pollutant emissions are taken from E-PRTR data.

Figure 1: The impact pathway approach as it relates to pollutant emissions



The purpose of the review was to highlight areas where new scientific developments could be taken into consideration and others where an update of methods or parameter values does not appear necessary or feasible. A further point to consider was the complexity of the proposed approach and how this may impact on the potential for regular updates of the data, balancing complexity and accuracy. The work was structured into two subtasks:

1. review and interim gap analysis,
2. preparation of final report.

The report resulting from the first subtask made suggestions for possible adjustments of the approach and explicitly formulated questions as part of a written consultation with a range of experts on the different steps of the impact pathway approach that took place in autumn 2019. The final working paper (the present document) takes account of the expert feedback.

The procedure followed in the review and the suggestions submitted are summarised below, separately for each step of the impact pathway approach.

Quantification of emissions

Pollutants reported in the previous assessment covering the years 2008-2012 (EEA, 2014) are the air pollutants ammonia (NH_3), nitrogen oxides (NOx), non-methane volatile organic compounds (NMVOCs), primary particulate matter ($PM_{2.5}$, PM_{10}) and sulphur oxide (SO_2), the heavy metals arsenic, cadmium, chromium, lead, mercury and nickel and the organic compounds 1,3 butadiene, benzene, dioxins/furans and polycyclic aromatic hydrocarbon.

In the current work, a screening of all E-PRTR pollutants for frequency of reporting was carried out. Additional screening was undertaken to analyse the relative toxicity impact of emissions reported to the E-PRTR, combining toxicity characterisation factors with mass of emissions to produce a toxicity impact score, based on characterisation factors from USEtox[1]. This screening identified 8 pollutants and 2 greenhouse gases that could be considered for inclusion in the updated assessment: carbon monoxide, copper, zinc, fluorine, chlorine, vinyl chloride, dichloromethane and 1,2-dichloroethane, methane and nitrous oxide.

Dispersion and exposure modelling

With respect to atmospheric transport (dispersion) and transformation and subsequent exposure modelling of the major air pollutants (mainly ozone, PM_{10}, $PM_{2.5}$ and NO_2 resulting from the emissions of NH_3, NOx, NMVOCs, $PM_{2.5}$, PM_{10} and SO_2), the EEA (2014) study relied on Source Receptor Matrices (SRMs) from the chemistry transport model EMEP. In a country-to-grid configuration, SRMs give the change in various pollution levels in each receptor grid resulting from a change in anthropogenic emissions from each individual country. Given that the height of industrial emission sources impacts on the dispersion and that industrial emissions may not be evenly distributed over the country, sectoral adjustments were applied. These adjustments were based on results available only for 4 countries.

Since the 2014 EEA report, improvements were achieved in modelling of secondary organic aerosols, and the model resolution for emissions used in the development of SRMs has increased. In the review, the ETC explored the use of other models (not available for the 2014 report) as a complement to EMEP. This was done in order to (i) explore the importance of non-linearity issues in impact responses due to emission changes (depending on the size of emission changes), (ii) assess alternative options for determining sectoral adjustments and extending them to all countries, as well as to (iii) assess the possibility of high resolution NO_2 modelling for health impact calculation. This work relied on tests using the Air Control Toolbox (ACT) from the Copernicus Atmosphere Monitoring Service (CAMS) and the SHERPA model developed by the Joint Research Centre (JRC).

<u>Major air pollutants</u>

Concerning the major air pollutants, we recommend the use of the most recent and validated EMEP Source Receptor Matrices (SRMs). Concerning the impact of ozone on agriculture, AOT40 SRMs will need to be used as long as PODy SRMs are not available.

For the assessment of NO_2 health impacts, the use of the CHIMERE-SHERPA model with 7km×7km grid-to-grid SRRs is feasible, with estimated errors associated to the grid resolution below 20% for no-threshold indicators. The current resolution of the CHIMERE-SHERPA tool (7km) still smoothens out NO_2 gradients, making it unfit to estimate population exposure to NO_2 concentration levels above a threshold (such as the 20 $\mu g.m^{-3}$ cut-off point as in the response function currently recommended by the WHO/Europe). A more spatially resolved CTM-SHERPA would be required to address this question.

[1] https://usetox.org/

Our suggestion is to use CHIMERE-SHERPA (and/or EMEP-SHERPA if available) with its current resolution for the 2020 update for response functions without cut-off point.

The desirability of deriving a unit damage costs (€/tonne of pollutant) specific to the industrial sector was identified in the EEA (2014) report and confirmed by the consulted experts. However, it is apparent that there is a lack of consensus on the best approach to dealing with this sector specificity. The results of the earlier EURODELTA II study are now outdated, so can no longer be regarded as state-of-the-art. Assessment carried out as part of this work finds that there are three options for proceeding:

1. Adopt average damage costs by country that do not take account of the specificity of the industrial sector. Given that the observed potency ratings from analysis so far using the SHERPA models are generally close to 1 (i.e. close to the national average damage cost), this position, though imperfect, may be a pragmatic solution to the problem.
2. Use the results of a specific model (SHERPA-EMEP or SHERPA-CHIMERE).
3. Use results averaged across SHERPA-EMEP and SHERPA-CHIMERE.

At the present time, limited results are available from the SHERPA models, especially regarding the temporal representativeness since only one year is targeted in the parametrisations. Further modelling is needed to better understand the way in which the results for different sectors vary across Europe. In particular, countries distant from the centre of Europe (e.g. Greece, Portugal, Finland for which the analysis has not yet been carried out) should be added to the analysis prior to making a final decision on the most appropriate approach and source for sector adjustments. Inclusion of these countries will improve understanding of the extent and reasons for variation. This work should be finalised by the end of Q1/2020, i.e. before the new methodology needs to be implemented.

Toxic metals and organic pollutants

Concerning dispersion and exposure of toxic metals and organic pollutants, for which not only inhalation but also ingestion is relevant, the Uniform World methodology was used for the 2014 assessment. It accounts for environmental accumulation, transport and estimation of concentrations in soil and water compartments, leading either to direct human exposure (e.g. by drinking water) or to movement of pollutants into the food chain. The work in 2019 reviewed available information on two further models, ESPREME[2] and Pangea[3] that could alternatively be used for future assessments. The analysis shows a variation in modelling frameworks, between simplified tools as used in EEA (2014) and more complex systems such as Pangea (Wannaz et al., 2018a, b, c). The consistency of the Pangea framework with existing European tools (EMEP, CHIMERE, etc.) for air pollution assessment is not known in detail. Consistency with other tools, however, seems good (e.g. EUSES, given the links with USEtox). It is also noted that complex models may not give clearly better results than simpler tools.

There is a balance to be struck between complexity, transparency and flexibility. Without systematic testing of the different tools it is not clear which are best suited to the current application. Perceived increase in accuracy of the exposure assessment for toxic metals and specific organic pollutants of relevance to the work under discussion needs to be seen in the wider context of the final estimates of damage per unit emission. Discussion in this working paper highlights greater uncertainties at the stage of defining which impacts should be quantified than seem likely to originate from the exposure assessment. We therefore conclude that exposure methods do not need to be updated compared to the 2014 assessment, while more focus should be given to the impact assessment. This position may be reviewed in the future, for example if the list of pollutants requiring quantification was to be extended.

[2] http://www.integrated-assessment.eu/eu/indexa62b.html?q=resource_centre/integrated_assessment_heavy_metal_releases_europe
[3] http://www.pangea-model.org.

Quantification of impacts

Concerning the major air pollutants, the EEA (2014) damage cost factors included:

- costs for health effects (mortality and morbidity) from exposure to fine particulate matter and ozone;
- corrosion of buildings from SO_2; and
- crop yield losses from ozone.

Health impacts were also calculated for exposure to toxic metals and organics. Costs for CO_2 were not based on damage assessments but on marginal costs of greenhouse gas (GHG) mitigation. Therefore, direct impacts of CO_2 were not assessed in the previous report, although carbon emissions were valued.

The review examined impacts of the various pollutants included in recent EU or national projects and guidance and publications of national health institutes or committees. It resulted in the following recommendations.

<u>Health</u>

- It is suggested to continue including damage costs for emissions of the major air pollutants that cover the impact of emission per tonne of pollutant from a particular country, wherever the impacts occur. We suggest to additionally present the damage occurring only in the country.
- There is some interest (for example from OECD) in splitting out the costs of air pollution to business (especially through impacts on productivity). These have always been included in the damage cost analysis for the EEA, though the methods used to date are to be reviewed and possibly updated by the end of February 2020. This active field of research will be kept under review up to the time that the models are run.
- For additional health effects included in recent analyses, but not in previous EEA assessments (such as cardio vascular disease or stroke), we will conduct a sensitivity assessment in 2020 but we consider inclusion in the best estimates at the present time as premature.
- There are a few publications suggesting updated (non-linear) response functions for mortality effects from $PM_{2.5}$. In the absence of a review of response functions by the WHO, we suggest continuing using the HRAPIE (WHO, 2013a) response functions for all-cause mortality. The possibility of a side evaluation (sensitivity calculation) with a more recent response function will be considered in the analysis in 2020.
- Health impacts from NO_2 were not included in the previous EEA assessment. We suggest extending future EEA assessments to include health impacts from NO_2. Morbidity impacts from NO_2 relying on response functions without cut-off points can be accounted for in the damage cost assessment. Concerning chronic mortality, however, different exposure-response functions are currently used (the most recent ones coming from COMEAP (2018)). Applying the response function with a cut-off point recommended by HRAPIE together with the available source-receptor matrices would involve high uncertainty and would tend to underestimate mortality effects from NO_2. Therefore, for the 2020 analysis we suggest the following. If the WHO proposes a new response function without a cut-off point, we will use this function. Otherwise, we will not include NO_2 chronic mortality into this year's damage cost assessment. However, a screening analysis to assess sensitivity of results to different available response functions for NO_2 chronic mortality on overall health damage will be carried out based on high resolution chemistry transport modelling in France. In this screening we will compare the HRAPIE function with the different COMEAP (2018) views.

- For trace metals and organic pollutants, recent analyses assess impacts not previously included in EEA assessments (e.g. mortality). Given, however, that these assessments are made for the birth cohort rather than for the population alive as is the case for other health impacts assessed in earlier EEA reports, plus the need to pay attention to the risk of double counting, for example when chronic mortality not only of PM but also of mercury and lead is taken into account, we conclude that further assessment is required before additional impacts of metals can be included in the analysis.

Materials

- We suggest extending the assessment of the impacts of air pollution on materials in future analyses and to include, on top of corrosion effects on materials, also impacts of soiling from particulate matter and impacts of ozone. Furthermore, information on updated response functions for materials that may have changed since the currently used functions were developed (e.g. paints) should be collected in the 2020 assessment.

Crops, forests and biodiversity

- The scientifically recommended indicator to assess impacts on crops and forests from ozone is the stomatal ozone flux. However, currently no PODy SRMs are available. For the time being, the calculation of impacts of ozone on crops and forests will hence need to continue using the AOT40 indicator as in the earlier EEA (2014) report. For the future we recommend the creation and publication of POD SRMs.
- Although uncertainty in quantifying ecosystems and biodiversity impacts is still high, we suggest calculating biodiversity effects from deposition of NH_3 and NO_2 for exceedances of critical loads in Natura 2000 areas. While monetised impacts are likely to remain low, the political importance of biodiversity, and the extent of critical loads exceedances for nitrogen, are high.

Greenhouse gases

- The consulted experts agreed with us that biomass use is not necessarily carbon neutral. The scientific debate on what assumptions to use with respect to the CO_2 emissions of biomass combustion is unresolved. We suggest applying a somewhat arbitrary middle option between assuming 0 and 100% carbon emissions.

Additional pollutants and non-CO_2 greenhouse gases

- With respect to pollutants whose integration was a suggestion from the screening analysis in Chapter 3, we are not aware of broadly agreed dose-response functions. Out of the 8 pollutants (carbon monoxide, copper, zinc, fluorine, chlorine, vinyl chloride, dichloromethane and 1,2-dichloroethane) and 2 greenhouse gases (methane and nitrous oxide) whose integration was suggested from the screening analysis, we suggest retaining only the two greenhouse gases. The contribution of methane to ozone, however, cannot be accounted for as there are currently no SRMs for methane.

Valuation of quantified impacts

Monetary values for different health or non-health end points affected by air pollution seek to capture associated costs in a way as comprehensive as possible. For health end points for example, they seek to capture costs to employers, costs of healthcare and willingness to pay to avoid pain, suffering and premature death. The general approach for valuation of impacts therefore combines individual willingness to pay for health (etc.) protection with additional elements, for example the costs of

healthcare and lost productivity. For building materials cost factors reflect repair and cleaning costs. For climate change, the preference in previous reports has been to base costs on the marginal cost of GHG mitigation.

The ongoing review investigated whether and which more recent unit costs, compared to the 2014 EEA study, are used for monetizing impacts from air pollutants and greenhouse gases. The recommendations of the review are the following.

Health

- Although cost data for morbidity impacts used in the 2014 assessment are rather dated, we suggest that any revision of the figures should be carried out in agreement with European Commission services to ensure that there are not inconsistencies in approach across the Commission and its agencies. In the 2020 report we will present recently updated morbidity costs for comparison with the applied values.
- The issue of whether to quantify mortality based on premature deaths (monetised by VSL), on years of life lost (monetised by VOLY), or on both, remains unresolved amongst scientists. This ongoing dispute was also reflected in the feedback we received during our expert consultation. We suggest continuing using both values in the upcoming EEA assessment and to present them as two alternative approaches.
- Cost data for mortality have been updated in the recent DG MOVE Transport cost handbook, basing their estimate on the OECD (2012) value for a VSL. The European Commission "Better Regulation Toolbox" both of 2015 and of 2017 makes reference to the same OECD value. We therefore suggest applying the same VSL estimate also in the 2020 EEA assessment.
- The DG MOVE transport handbook applies a VOLY of 70 k€. According to personal communication, it is likely to be used also in the ongoing DG ENV study "Mapping objectives in the field of environmental taxation and budgetary reform: Internalisation of environmental external costs". Whatever value is adopted for the VOLY should be checked for consistency with the adopted VSL estimate.

Materials

- There has been no substantial new work on materials damage. We suggest updating previous cost data for repair or replacement of materials in line with inflation.

Crops, forests and biodiversity

- With respect to valuing damage from ozone on crops and forests we do not suggest any change to the valuation. The simple approach of using producer prices, country specific or European average, appears justifiable given the limited contribution of this endpoint to the overall damage costs.
- Effects on ecosystems and biodiversity were not accounted for in the 2014 assessment. New assessments are available. Although estimated economic damage remains small compared to health damage from air pollution, given the political and public interest in biodiversity losses, we suggested to include such effects into the assessment. We suggest basing the assessment in 2020 on willingness to pay (WTP) values.

Greenhouse gases

- Concerning carbon valuation, the use of damage costs for carbon valuation would be more in line with the approach applied to the valuation of health effects and present the scientifically sound approach. Review of current methods shows that most EU countries that publish guidance apply the marginal abatement cost approach. An exception is Germany. We are not aware of any harmonised EC guidance on carbon valuation. Nevertheless, DG MOVE in its

transport cost handbook promotes a marginal abatement cost approach. Communication with CE Delft and Trinomics involved in the ongoing studies for DG ENV and DG ENER suggest they are likely to follow the approach used in the DG MOVE Transport Cost Handbook. Chosen values should ideally be accepted by DG CLIMA. In the absence of further guidance from the EC we suggest following the approach and values applied in the DG MOVE study.

Acknowledgements

The EEA task manager was Ian Marnane in 2019, and Bastian Zeiger in early 2020.

We thank the numerous experts who have provided comments on the first version of this working paper. Their feedback has clearly helped to clarify and improve this working paper.

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1 Introduction

1.1 Context

The EEA has published two reports assessing the costs of air pollution from EU industrial facilities: the report "Revealing the costs of air pollution from industrial facilities in Europe" published in 2011, and the report "Costs of air pollution from European industrial facilities 2008–2012, an updated assessment" published in 2014 (EEA, 2011 & 2014).

These reports were carried out based on best practice at the time, with the 2014 report presenting an updated assessment of the 2011 report. The EEA is considering a further update on this assessment in 2020. In advance of initiating the assessment the EEA identified a need to critically review the methodology used in the 2014 report and identify areas where improvements could be made to further strengthen the outputs from the assessment.

The EEA invited its European Topic Centre on Air pollution, noise, transport and industrial pollution (ETC/ATNI) to carry out this review in 2019. The team involves INERIS and Aether with EMRC as an external consultant. The ETC/ATNI partner NILU is involved as internal reviewer.

1.2 Objectives

The review was to cover all the principle steps of the impact pathway approach (IPA) (ExternE, 2005), from emissions through exposure to the quantification of health and environmental impacts and their monetary valuation, as well as the consideration of uncertainties. It was to highlight areas where new scientific developments could be taken into consideration and others where an update of methods and parameter values does not appear necessary or feasible. A further point to consider was the complexity of the proposed approach and how this may impact on the potential for regular updates of the data[4], balancing complexity and accuracy. The suggested methods are to be robust, capable of highlighting changes over time and their origins, and easy to repeat regularly.

The work was structured into two subtasks: "Review and interim gap analysis" (subtask 1) and "Preparation of report" (subtask 2).

Subtask 1 consisted of a detailed review of the 2014 report, reviewing all elements of the impact pathway assessment and considering current best practice as presented by results of EC projects, Commission, WHO and/or national guidance or publications of national health institutes or committees. A review of individual epidemiological publications is outside the scope of this report. The assessment considered also the basic aspects of the overall scope of the study, including pollutants, geographical extent, scope of receptors to be considered and timescale under consideration. Where areas for improvement were identified, initial changes to the methodology were considered to address these areas.

The first deliverable was a preliminary version of the final ETC Working Paper focusing on the gap analysis. It was provided to the EEA for review and discussion and submitted to a written consultation and discussion by stakeholders. A summary was furthermore sent to the EIONET network for comments.

Based on the comments received, the ETC prepared a more comprehensive ETC Working Paper under *subtask 2* (the present version). This working paper identifies areas of weakness, uncertainty or areas

[4] This could involve updating the calculation of externalities of industrial facilities annually or biennially, based on the most recent emissions data, including a very brief assessment of any significant recent research or published methodologies which could indicate that the overall methodology should be updated. More irregular updates of the methodology would be carried out every 5 or so years and when fundamental issues with the assessment methodology arise.

where scientific knowledge and available tools have improved since the completion of the 2014 assessment. It outlines the proposed approaches and the reasons for changes compared to the 2014 report. It also identifies areas where more analysis is necessary before making final decisions.

1.3 Scope of this report

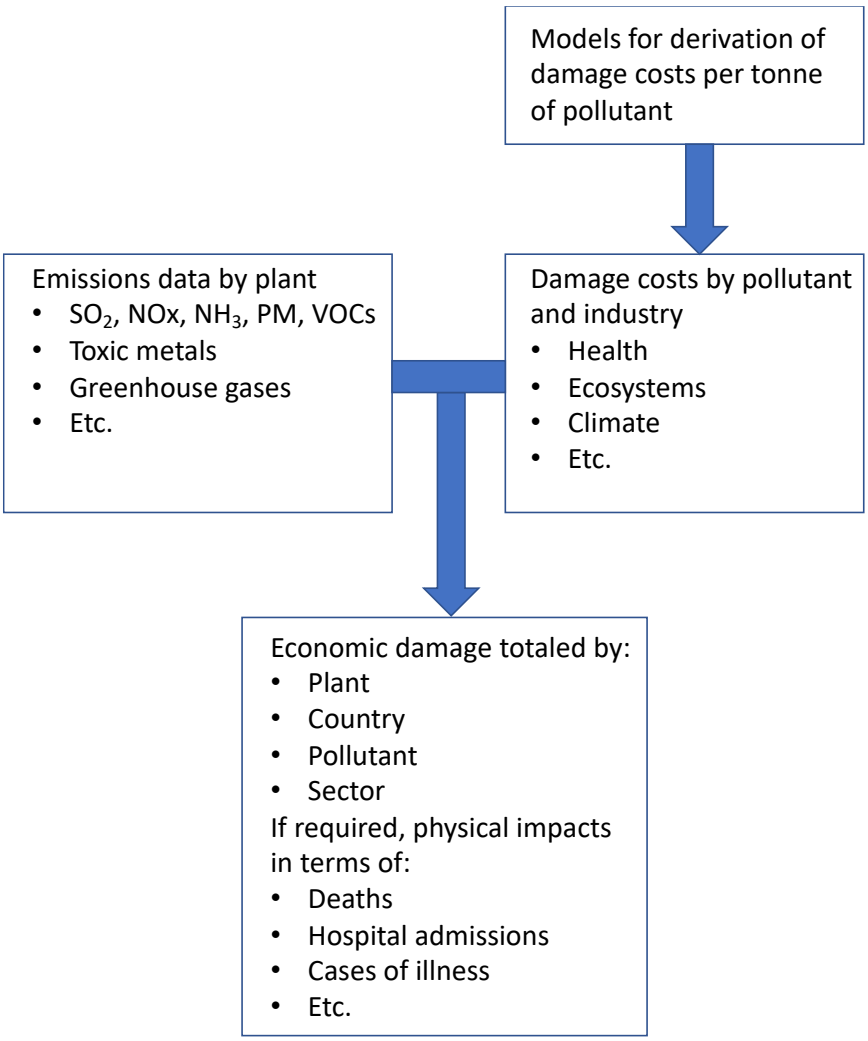
This report is the deliverable under subtask 2. It focuses on the presentation of previous assessments and on the identification of gaps and new scientific developments. The report is structured as follows. Chapter 2 recalls the origins and presents the framework of the impact pathway approach and of the overall approach to quantifying externalities. Chapter 3 focuses on emission data, shortcomings in the E-PRTR data base identified in previous assessments and recent developments, as well as a pollutant screening aimed at reviewing the scope of the substances covered. Chapters 4 to 6 review the methods of the previous assessment report with respect to exposure assessment, impact quantification and monetary evaluation, respectively. For each subject scientific development since the 2014 report is presented indicating potential gaps in the 2014 approach and areas where improvements could be considered. The conclusions in each chapter summarise the potential gaps and areas where a revision of data and approaches is suggested. They take account of the feedback received during the expert consultation.

2 The framework for quantifying externalities

2.1 The overall framework for analysis

The approach for quantifying externalities used in EEA (2014) is shown in outline in Figure 2. The key inputs to the analysis are data on emissions, taken from the E-PRTR, and unit damage costs per tonne emission, specific to industrial facilities. Multiplying emission by unit damage cost provides the estimate of economic damage linked to the release of a pollutant. With those two types of input it is possible to calculate a variety of damage estimates, as indicated in the figure.

Figure 2: Outline for quantifying externalities of industrial plant



Much of the rest of this report concerns the models for derivation of the unit damage costs (top right in the figure), given that the E-PRTR data are already available and the combination of E-PRTR emissions with damage costs is straightforward.

2.2 The impact pathway approach for deriving damage per tonne estimates

The impact pathway approach (IPA) was developed in functional form under the ExternE (Externalities of Energy) study funded by the European Commission in the early 1990s (ExternE, 1995, 1997, 2005). The externalities, or external costs, referred to in ExternE are effects on third parties arising from an

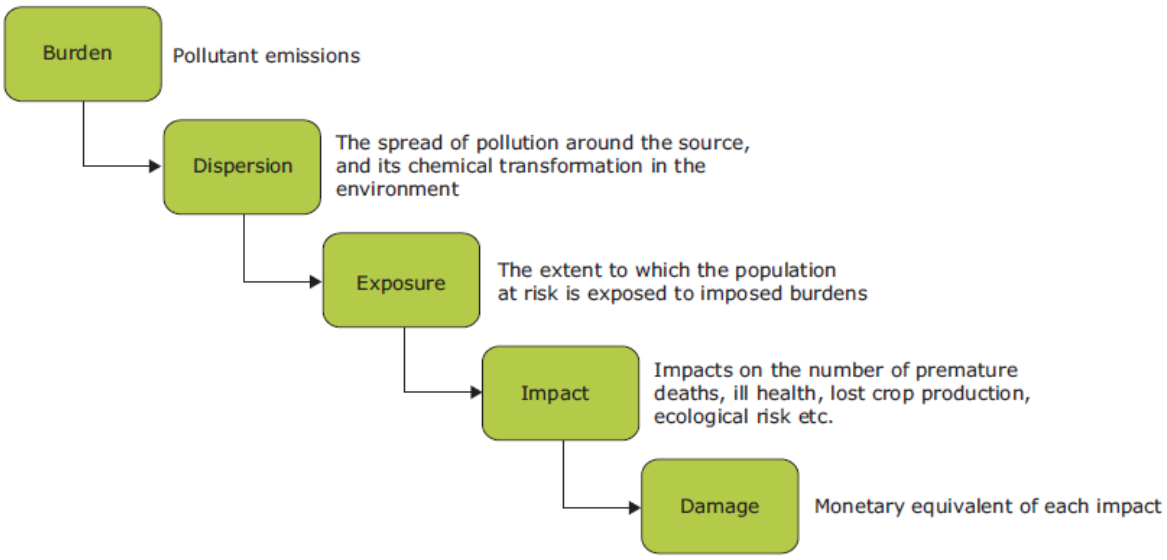
activity that are not accounted for by those undertaking the activity. Hence for the development of a coal-fired power station, air pollution externalities include damage to human health, ecosystems and building materials.

The framework developed in ExternE was developed to be able to address in a consistent manner that made use of the latest available scientific information any pressure that would generate external costs. As such it was designed to address damage from occupational disease and accidents, noise, visual intrusion of industrial plant, water pollution and various other stresses. In all cases it provides a simple logical progression from the generation of a burden (e.g. increased risk of accidents, or pollutant emission) through exposure of sensitive receptors (people, ecosystems, buildings, etc.) to the burden, quantification of impact and finally valuation.

Prior to the 1990s the ability to implement the IPA was constrained by limits in computational time and data availability. Earlier (but also some more recent) estimates of external cost of pollution were largely based on top down approaches (e.g. Hohmeyer, 1988; Bachmann, 2015; Pearce et al., 1992) where the characterisation of the link between (in the case of air pollution) emission and impact was characterised to only a limited degree.

An overview of the IPA for pollutant emissions is shown in Figure 3. It shows a logical progression from emission to monetary valuation, through pollutant dispersion and transformation, exposure of receptors including people, materials and ecosystems, impact quantification and the translation of damage into monetary value. Whilst the overall framework for analysis has not changed for over 25 years, the inputs to the modelling have been revised as knowledge of pollutant emission, exposure, effects and valuation has grown.

Figure 3: The impact pathway approach as it relates to pollutant emissions

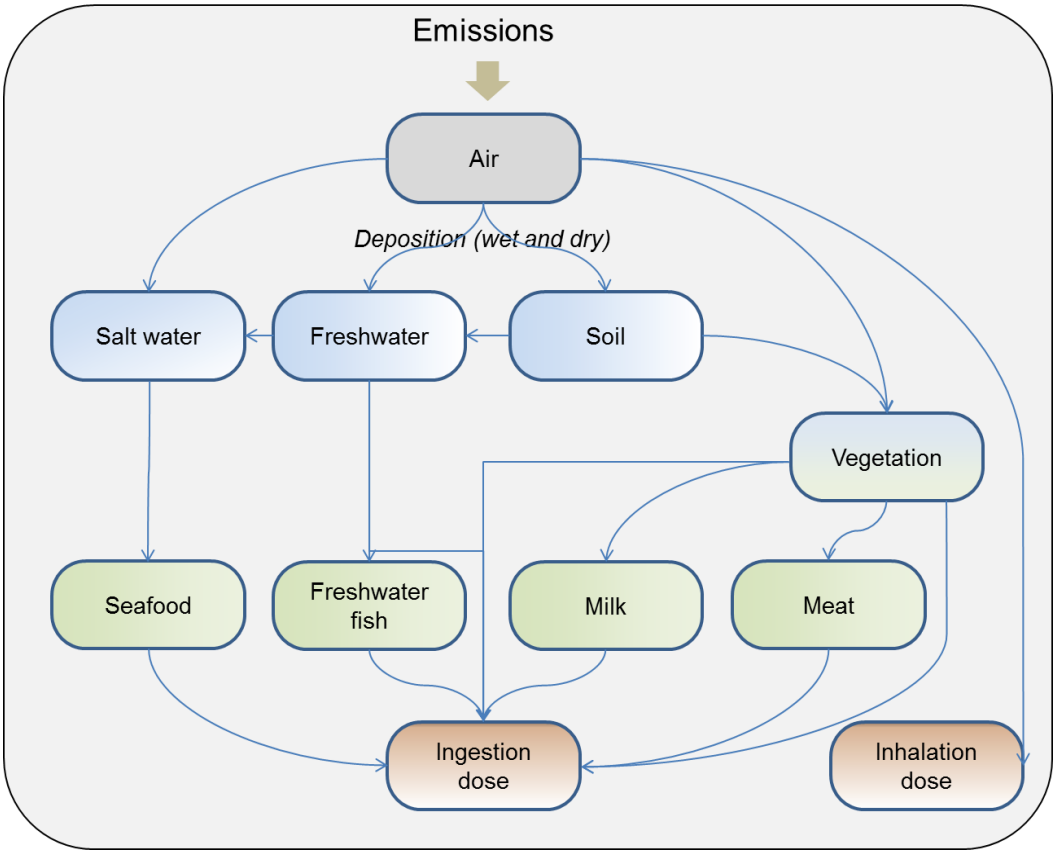


Historically, the IPA has been used most extensively in characterisation of air pollutant damages for example in the context of developing the emission ceilings directive or air quality directives (e.g. Holland, 2014). In recent years, socio-economic assessments for chemicals have used the IPA in relation to analysis carried out under the EU's REACH (Registration, Evaluation, Authorisation and Restriction of Chemicals) Regulation following guidance provided by ECHA (2011). Further examples exist, for example in relation to assessment of pesticides (Fantke et al., 2012)

The precise form of the IPA varies from pollutant to pollutant. In order of increasing modelling complexity these are:

- Unreactive fine particles, and some metals and organics for which risk is assessed against inhalation only (least complex) where exposure is modelled against the concentration of the pollutant which stays in the form in which it is emitted.
- Reactive pollutants such as SO_2, NH_3, NOx and VOCs for which conversion to secondary aerosol and ozone needs to be modelled.
- Some metals and organics where risk is associated with ingestion as well as inhalation, and for which flows through the environment to food, water and milk may need to be modelled (Figure 4, complex, showing the pathways modelled in the EEA (2014) report which focused on human exposure via emissions to air).

Figure 4: The impact pathway approach: pathways for exposure, as applied in EEA (2014).



As shown in later parts of this report (Chapters 5 and **6**), the IPA approach is applied widely already[5]. It is a simple, logical and sequential description of the evolution of impact following release of a pollutant and can integrate the latest scientific data. We therefore support the continued use of the approach, to the extent possible, also in future EEA damage cost assessments.

The IPA has not been used so extensively in policy development and appraisal of global problems, for example in relation to the release of greenhouse gases (GHGs) and ozone depleting substances. For the ozone depleting substances, impact assessment beyond modelling effects on the ozone layer, and of its recovery, is not well developed, though there are some examples (see Corden et al., 2017,

[5] It is the generally favoured approach, at least in terms of informing, directing and supporting EU Commission policy measures.

chapter 13). This provides an interesting case where the threat to health was considered so clear that it was possible to develop a truly global response without recourse to a detailed benefits assessment. However, more important to the current assessment is modelling of the economic damage linked to climate change. Here, the issue is different, given the presence of significant uncertainties in the modelling of impacts, for example in relation to the size and wealth of the future global population, its ability to adapt to a changing climate, and emission rates. As an example, Dong et al. (2019) find an order of magnitude difference in GHG damage costs. Auffhammer (2018) reviews the current state of science, albeit with a particular focus on the damage costs used in the USA, and finds significant deficiencies, both in the costs that are recommended and the studies that feed into them. The complexity of this modelling and associated uncertainties have caused alternative approaches to be considered. A number of European studies including EEA (2014) have applied marginal abatement costs for valuation of the GHG emissions rather than damage costs although this raises questions of consistency for the overall assessment of damage. These issues are explored in more detail in chapters 5 and 6.

3 Quantification of emissions - E-PRTR

The EEA externalities assessment aiming at quantifying costs of air pollution from European industrial facilities relies on emission data reported under the European Pollutant Release and Transfer Register (E-PRTR). The E-PRTR contains information on 91 pollutants including greenhouse gases, ozone-depleting substances, heavy metals, pesticides, acidification precursors and persistent organic pollutants. The choice of polluting substances and categories of substances reported to the E-PRTR relies primarily on the list of substances in the Kiev Protocol. There is however significant overlap with a range of different legal texts including: the EU Water Framework Directive, the United Nations Framework Convention on Climate Change, the Stockholm Convention on Persistent Organic Pollutants (POPs), the Rotterdam Convention on the Prior Informed Consent Procedure for Certain Hazardous Chemicals and Pesticides in International Trade, the Convention for the Protection of the Marine Environment of the North-East Atlantic, the International Convention for the Prevention of Pollution from Ships, and the UNECE Convention on Long-Range Transboundary Air Pollution, and also the earlier European Pollutant Emission Register (EPER) list of substances (UNECE, 2008).

E-PRTR data cover reporting from 2007 to 2017 by EU Member States, Iceland, Liechtenstein, Norway, Serbia and Switzerland6.

This section summarises the evolution of the register, recalls the polluting substances covered in previous EEA cost assessments and presents the results of a pollutant screening within the E-PRTR, identifying additional pollutants that could be considered for inclusion in the assessment.

3.1 Evolution of the Pollutant Release and Transport Register

While representing a useful resource for emission data, the E-PRTR is subject to different uncertainties. Not all facilities must report their emissions to the E-PRTR and not for all polluting substances included in the register. Indeed, the E-PRTR Regulation (EU, 2006) defines the industrial sectors that must report information to the register, as well as the pollutant- and activity-specific reporting thresholds. Only facilities exceeding these thresholds must report information to the register. In the past (EEA, 2011), it was suggested that there were important variations in completeness of reporting (omissions obviously biasing costing results), and that more extensive data checks could help improve data quality. Another issue has been a difficulty in tracing facilities over time, when facilities change ownership, name and/or national facility identification code or where locational references change over time, from a village location to the nearest town or district for example.

The method by which pollutant releases to air are quantified can have a significant effect on the uncertainty of the values reported to the E-PRTR, as well as simply on whether releases are determined to be above the reporting threshold. There are three options available to operators to report on the method used to quantify a release: measurement, calculation or estimation, along with a range of specific methodologies that can be used. Currently there can be significant heterogeneity in which method types and methodologies are reported as used to quantify releases of some pollutants. Improved guidance to operators and competent authorities would help to provide more consistency and comparability.

Incomplete reporting of some or all pollutants by some facilities has also been identified in the 2014 update of the assessment report (EEA, 2014). Furthermore, it was suggested that additional information on fuel consumption or production data would not only enable to assess a facilities' environmental impacts relative to its output, but also help in identifying potential reporting errors.

[6] It could also be considered whether damage cost data should be generated for countries at the borders of the EU (Turkey, western Russia, Ukraine, Belarus and countries in the Balkan region) to facilitate similar calculations being made for those countries, given their influence on air quality within the EU. The possibility to obtain source receptor matrices for all of these would have to be assessed.

Work was completed in 2017 to clean up the E-PRTR database and develop a more reliable and consistent timeline of emissions for individual facilities. The introduction in 2019 of the new EU Registry on Industrial Sites and integrated E-PRTR and LCP reporting should improve the completeness and tracking of facilities with the introduction of static IDs for industrial entities and separation of the emission and facility data flows. Under the EU Registry on Industrial Sites all E-PRTR facilities above the activity thresholds must be reported, regardless of actual emission amounts. New automated QA checks are also hoped to improve the reliability of E-PRTR reporting, with feedback on the data being provided before submission. In addition, the recent Commission Implementing Decision has made reporting production volume data to the E-PRTR mandatory from 2021. It has however been found, through Weibull analysis, that for the majority of pollutants relevant in the context of this study, releases to air reported to the E-PRTR cover between 80 and 90% of total industrial emissions, with the exception of emissions of ammonia.

The previous assessment (EEA, 2014) covered 29 countries, the 27 EU Member States[7], Norway and Switzerland. Iceland was not included in the transfer matrices derived for the GAINS model using the EMEP model, and CO_2 emissions for Serbian facilities were not available. These countries, therefore, were not included in the analysis. E-PRTR data currently cover the 28 EU Member States, Iceland, Liechtenstein, Norway, Serbia and Switzerland.

3.2 Screening of pollutants – frequency of reporting

Pollutants reported in the previous assessment covering the years 2008-2012 (EEA, 2014) are the air pollutants ammonia (NH_3), nitrogen oxides (NOx), non-methane volatile organic compounds (NMVOCs), particulate matter ($PM_{2.5}$, PM_{10}) and sulphur oxide (SO_2), the heavy metals arsenic, cadmium, chromium, lead, mercury and nickel and the organic compounds 1,3-butadiene, benzene, dioxins/furans and polycyclic aromatic hydrocarbon (as BaP equivalent). However, 1,3-butadiene is not a pollutant reported to the E-PRTR and has therefore been excluded from this screening analysis.

The reporting frequency of the pollutants investigated in the previous EEA report has been summarised for the years 2012-2016 (Table 1). Reporting of all these pollutants is high, except chromium, with the majority of countries reporting releases at least once and usually every year.

Reported emissions from Serbia are still low, as found in the previous report no CO_2 emissions were reported between 2012-16.

[7] Excluding Croatia.

Table 1: Number of years each pollutant was reported by country 2012 - 2016

Country	Number of years pollutant was reported														
	Ammonia	Arsenic	Benzene	Cadmium	Carbon dioxide	Chromium	Lead	Mercury	Nickel	Nitrogen oxides	Non-methane volatile organic compounds	PM_{10}	Dioxins + Furans	PAH	Sulphur oxides
Austria	5	2	3	5	5	2	5	5	5	5	5	5	4	2	5
Belgium	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Bulgaria	5	3	1	5	5		5	4	1	5	5	5	1		5
Croatia	3	3		3	3			3		3	3	3	1		3
Cyprus	5	5		5	5			5	5	5		5			5
Czech Republic	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Denmark	5	5	1		5			5	1	5	5	5	4		5
Estonia	5	5	3	5	5	5	5	5	5	5	5	5	2		5
Finland	5	5	5	5	5	5	5	5	5	5	5	5	5	4	5
France	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Germany	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Greece	5	5	5	5	5	5	5	5	5	5	5	5	4	3	5
Hungary	5	1	4		5		5	5		5	5	5	1	4	5
Iceland	5				5							5	3	5	5
Ireland	5	5			5	1		5	5	5		5	1	5	5
Italy	5	5	5	5	5	5	5	5	5	5	5	5	5	4	5
Latvia	5			1	5			3		5	5	5			4
Lithuania	5		5		5	3	1			5	5	5			5
Luxembourg	3		3	2	5	2	1	5		5			5	1	5
Malta	1	5		4	3				5	5		5	1		5
Netherlands	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Norway	5	5		4	5	4	5	4	5	5	5	5	5	2	5
Poland	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Portugal	5	5	5	5	5	4	5	5	5	5	5	5	5	5	5
Romania	5	4	2	5	5	5	3	5	5	5	5	5	5	3	5
Serbia	5									5	3	5			5
Slovakia	5	5	5	5	5	5	5	5	5	5	5		4		5
Slovenia	5	2	5	3	5	5	5	5	5	5	5	5	1		5
Spain	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Sweden	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Switzerland	5		5	1	5		5	5	5	5	5		1	5	5
United Kingdom	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5

This reporting frequency is comparable to the years (2008-2012) that were covered in the previous report (EEA, 2014, see Table 2). The exception is for chromium and compounds, which were reported more frequently over the earlier timeframe compared to the more recent one.

Table 2: Number of years each pollutant was reported by country 2008 – 2012 (used in 2014 report)

Country	Number of years pollutant was reported														
	Ammonia	Arsenic	Benzene	Cadmium	Carbon dioxide	Chromium	Lead	Mercury	Nitrogen oxides	Nickel	Non-methane volatile organic compounds	PM_{10}	Dioxins + Furans	PAH	Sulphur oxides
Austria	5		3	5	4	5	5	5	5	5	5	5	3	5	5
Belgium	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Bulgaria	5	4	5	5	2	5	3	5	4	5	5	2		5	5
Cyprus	5	5	5	5			5	5	5	2	5			5	5
Czech Republic	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Denmark	5	4		5			5	5	2	5	4	3		5	5
Estonia	5	5	5	5	5	5	5	5	5	5	5	1		5	5
Finland	5	5	5	5	5	5	5	5	5	5	5	5	3	5	5
France	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Germany	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Greece	5	5	5	5	5	5	5	5	5	5	5	4	1	5	5
Hungary	5	4	2	5	1	5	5	5	2	5	5	4	5	5	5
Iceland	5	1		5				5			5	3	5	5	5
Ireland	5	5	2	5	3	1	5	5	5	3	5	2	3	5	5
Italy	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Latvia	5			5			3	5		5	4		2	5	5
Lithuania	5			5				5	1	5	5			5	5
Luxembourg	3	2	3	5	2	2	4	5	1	3	1	5		5	3
Malta		5	5	5				5	5		5			5	
Netherlands	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Norway	5	5	5	5	2	5	5	5	5	5	5	5	3	5	5
Poland	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Portugal	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Romania	5	3	5	5	4	4	5	5	5	5	5	5		5	5
Serbia	2									2	4			4	2
Slovakia	5	5	5	5	5	5	5	5	5	5		1		5	5
Slovenia	5	5		5	5	2	5	5	5	5	5	5	2	5	5
Spain	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Sweden	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5
Switzerland	5	3	4	5	3	5	5	4	4	5		2	5	5	5
United Kingdom	5	5	5	5	5	5	5	5	5	5	5	5	5	5	5

Note Serbia and Iceland were not included in the report (transfer matrixes were not available for Iceland and there were no CO_2 emissions from Serbia)

Additionally, the reporting frequency has not changed significantly for most of the pollutants in the previous report and for the majority of other air pollutants. Table 3 shows the reporting frequency for all other air pollutants not included in the 2014 EEA report over the 2012-2016 timeframe. A majority of the organic pollutants remain infrequently reported. Pollutants reported with a frequency comparable to those that were previously investigated are two greenhouse gasses: methane and nitrous oxide, and five air quality pollutants: carbon monoxide, copper and compounds, fluorine and inorganic compounds, chlorine and inorganic compounds and zinc and compounds. It therefore may be beneficial to consider including these pollutants in future analyses.

The gaseous pollutants are among the highest pollutants, by kg, reported to the E-PRTR. While the emissions of copper and zinc that are reported are less than many pollutants to air, they are reported on an equivalent frequency to the other metals included in the previous report.

Table 3: Number of years each pollutant was reported by country 2012 – 2016 of all pollutants reported to the E-PRTR (excluding those where there was only one instance of it being reported)

Country	1,1,1-trichloroethane	1,1,2,2-tetrachloroethane	1,2-dichloroethane (DCE)	Aldrin	Anthracene	Asbestos	Carbon monoxide (CO)	Chlorine and inorganic compounds	Chlorofluorocarbons	Copper	Di-(2-ethyl hexyl) phthalate	Dichloromethane	Ethylene oxide	Fluoranthene	Fluorine and inorganic compounds	Halons	Hexachlorobenzene	Hydrochlorofluorocarbons	Hydro-fluorocarbons	Hydrogen cyanide	Methane	Naphthalene	Nitrous oxide	Perfluorocarbons	Phenols	Polychlorinated biphenyls	Polycyclic aromatic hydrocarbons	Sulphur hexafluoride	Tetrachloroethylene (PER)	Tetrachloromethane (TCM)	Trichlorobenzenes (TCBs) (all isomers)	Trichloroethylene	Trichloromethane	Vinyl chloride	Zinc and compounds (as Zn)
Austria			5				5	5		5		4			3			3	5	5	5	2	5				2								5
Belgium	2		5		3		5	5	5	5	5	5			5	5		5	5	5	5	5	5	5		5	5	5	5	5			5	5	5
Bulgaria							5			5											5		5												5
Croatia							3																3												3
Cyprus							5			4											5		5												5
Czech Republic			5				5	5	2	5	5	5			5			5	5	4	5	1	5			5	5	1	4			5		4	5
Denmark							5	5							5			5	4	2	5		5			3									2
Estonia							5	2		5										2	5														5
Finland							5	5		5		4			5		3		2	3	5	5	5			5	4								5
France	5	5	5		5		5	5	5	5		5	5		5	5	1	5	5	5	5	5	5	5			5	5	5	5	1	4	5	5	5
Germany			5				5	5	5	5		5			5	1	4	5	5	5	5	5	5	5		5	5	4		5	1		5	5	5
Greece		5					5	5	5	5	5	5			5			5	5	5	5	1	5	5		3	3		5	3	3				5
Hungary							5	5		5		5			1					5	5		5				4		2				2	5	5
Iceland							5								5						5			5			5								
Ireland							5	5	5			4						5	5		5		5	4			5	4							
Italy	1		2		2		5	5	5	5	5	5			5			5	5	5	5	5	5	5		5	4	5	4	1		5	2	1	5

Country	1,1,1-trichloroethane	1,1,2,2-tetrachloroethane	1,2-dichloroethane (DCE)	Aldrin	Anthracene	Asbestos	Carbon monoxide (CO)	Chlorine and inorganic compounds	Chlorofluorocarbons	Copper	Di-(2-ethyl hexyl) phthalate	Dichloromethane	Ethylene oxide	Fluoranthene	Fluorine and inorganic compounds	Halons	Hexachlorobenzene	Hydrochlorofluorocarbons	Hydro-fluorocarbons	Hydrogen cyanide	Methane	Naphthalene	Nitrous oxide	Perfluorocarbons	Phenols	Polychlorinated biphenyls	Polycyclic aromatic hydrocarbons	Sulphur hexafluoride	Tetrachloroethylene (PER)	Tetrachloromethane (TCM)	Trichlorobenzenes (TCBs) (all isomers)	Trichloroethylene	Trichloromethane	Vinyl chloride	Zinc and compounds (as Zn)
Latvia							5	2				5			3						5		5										5		
Lithuania							5			5					5				2		5		5						4						5
Luxembourg							5			2											5	5				5	1								5
Malta																					5														5
Netherlands		1	4		2		5	5	5	5		5	4		5	2		5	5		5	5	5	3			5			5		5	2	5	5
Norway			5		5	3	4			5		2		5	3			3			5	4	5	5	5	1	2			3		1	1	5	5
Poland			5	1	2		5	5	5	5	5	5	2		5	1		5	5	5	5	3	5			1	5		5			1	5	5	5
Portugal		1					5	5		5		4			5			5	5	5	5	5	5				5		5		1		5	5	5
Romania							5	3		5										1	5		5	5		3	3								5
Serbia																					2														
Slovakia			5				5	5		5					5				5	5	5	5	2											5	5
Slovenia							5	5		2					5			3	5		5	3	5	5											
Spain		4	5		5		5	5	5	5	2	5			5			5	5	5	5	5	5	5		5	5	3	4	5		4	5	5	5
Sweden			5				5	5	5	5					5			3	5		5	5	5	5			5	4					4	5	5
Switzerland							3	4	4			5				4		4	5		5	2	5			5	5	5	2			3	1		5
United Kingdom	5	3	5		5		5	5	5	5	4	5	4		5	5		5	5	5	5	5	5	5	3	5	5	5	3	5	4	3	5	5	5

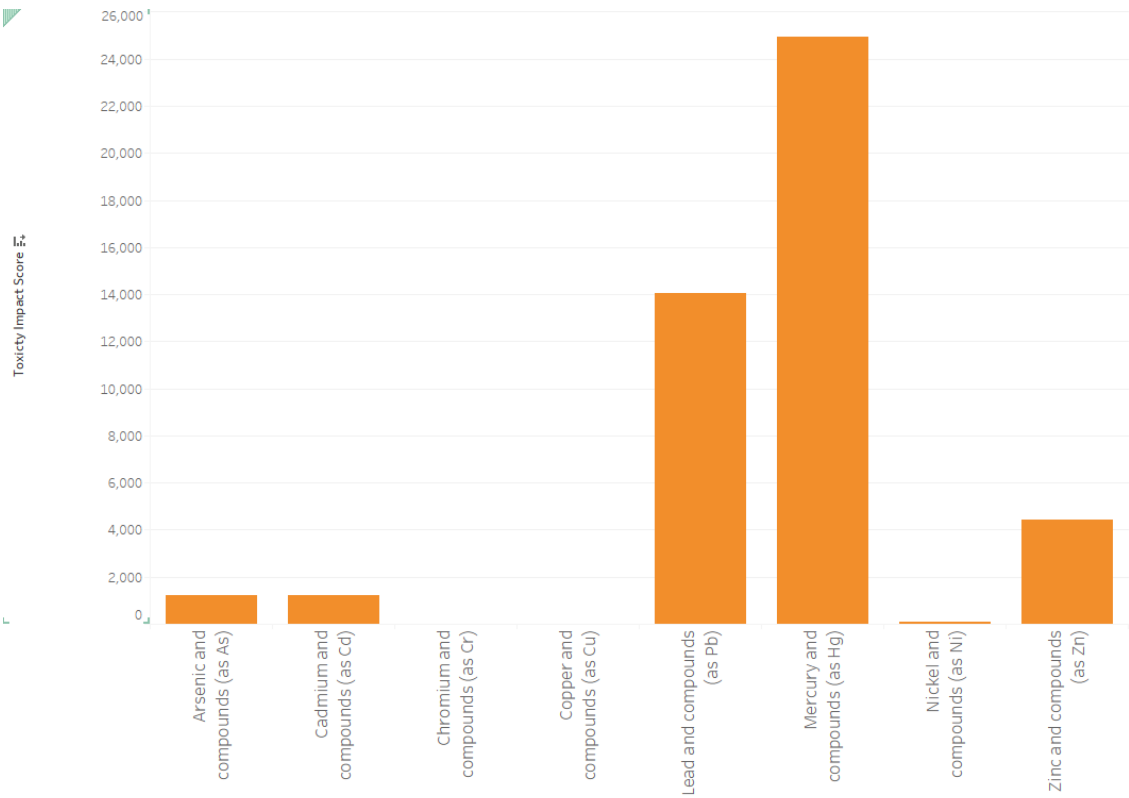
Note: Pollutants with high reporting frequency not used in the previous report are highlighted.

3.3 Screening of pollutants – human toxicity and ecotoxicity

Additional screening was undertaken to analyse the relative toxicity impact of emissions reported to the E-PRTR. This was not undertaken to identify every chemical substance that might be of potential interest but to identify any further pollutants reported under the E-PRTR that may be of interest for the proposed externalities assessment. For this we chose to use USEtox[8], combining toxicity Characterisation Factors (CFs) for emissions to air with mass of emissions to produce a toxicity impact score, in a similar manner to the methodology used in the recent Indicators for Industrial Emissions Policy report[9]. The endpoint human toxicity characterization factors are for emissions to continental rural air in units of disability adjusted life years (DALY) per kg emitted. The end point ecotoxicity characterisation factors are for emissions to continental air affecting freshwater ecosystems in units of potentially disappeared fraction (PDF) of freshwater species integrated over exposed volume and time (m^3.day) per kg emitted. For the metallic pollutants, as reported only on a very high level (e.g. arsenic and compounds), the Characterisation Factors for the most common oxidation state was used (e.g. As (III)).

The resulting relative toxicity impact scores are split into effects on humans and effects on the freshwater ecosystems; however, it should be noted that USEtox CFs were not available for all pollutants. The analysis (Figure 5 and Figure 6) showed that all the metallic air pollutants posed a significant risk to the environment and human health and supports the inclusion of copper and zinc in future analysis.

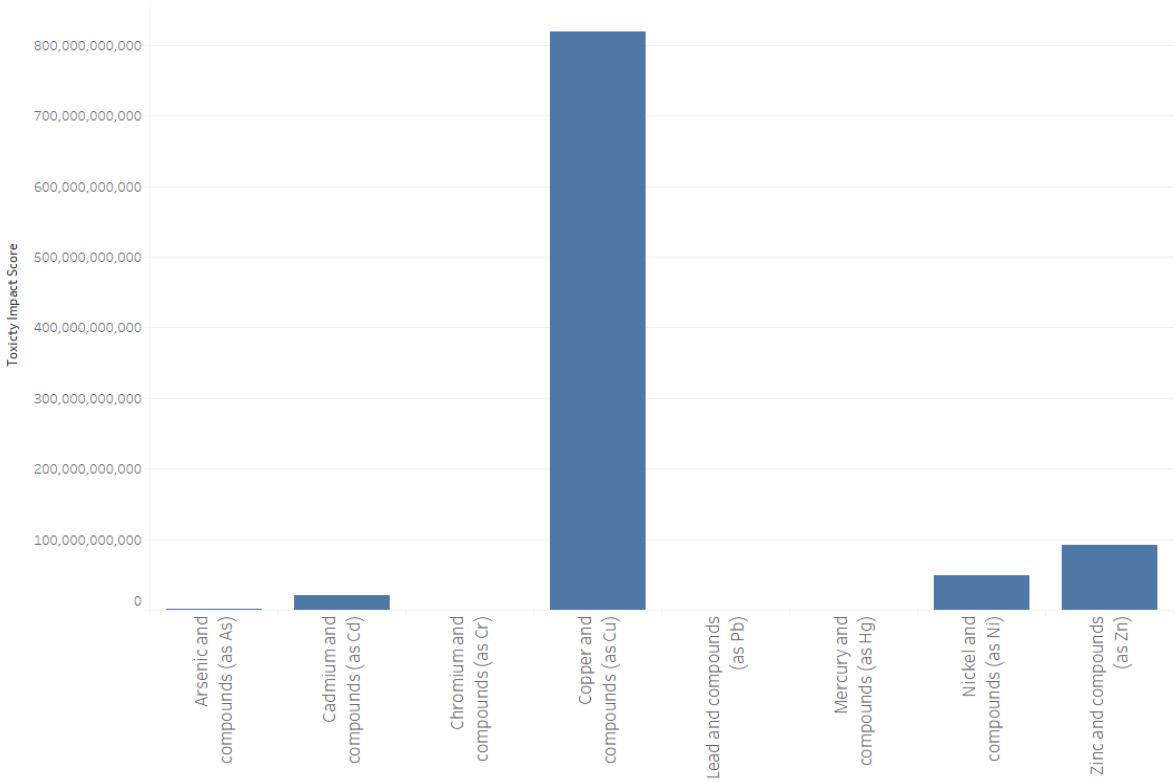
Figure 5: Average human toxicity impact scores of the metallic air pollutants reported to the E-PRTR between 2012-16



[8] https://usetox.org/

[9] https://circabc.europa.eu/w/browse/523fc5ff-fb45-4ff3-a513-9ee7268e967b

Figure 6: Average ecotoxicity impact scores of the metallic air pollutants reported to the E-PRTR between 2012- 16



Not all the non-metallic air pollutants identified as of possible interest with respect to the frequency of emission reporting (see Table 3) have toxicity data available through USEtox. Figure 7 and Figure 8 show the ecotoxicity and human toxicity impact scores for the non-metallic pollutants reported to the E-PRTR with USEtox CF coverage. It was found that ecotoxicity impact scores were relatively low for the majority of the organic pollutants. However, comparing against the toxicity impact score of benzene, the analysis identified 10 pollutants of interest (Table 4, out of which three may be worth including in future analysis: vinyl chloride, dichloromethane and 1,2-dichloroethane. It should be noted that aldrin has been excluded from this analysis as the large quantity reported between 2012 and 2016 was just one large, presumably erroneously reported, release in 2016.

Figure 7: Average human toxicity of the non-medallic air pollutants reported to the E-PRTR between 2012-16

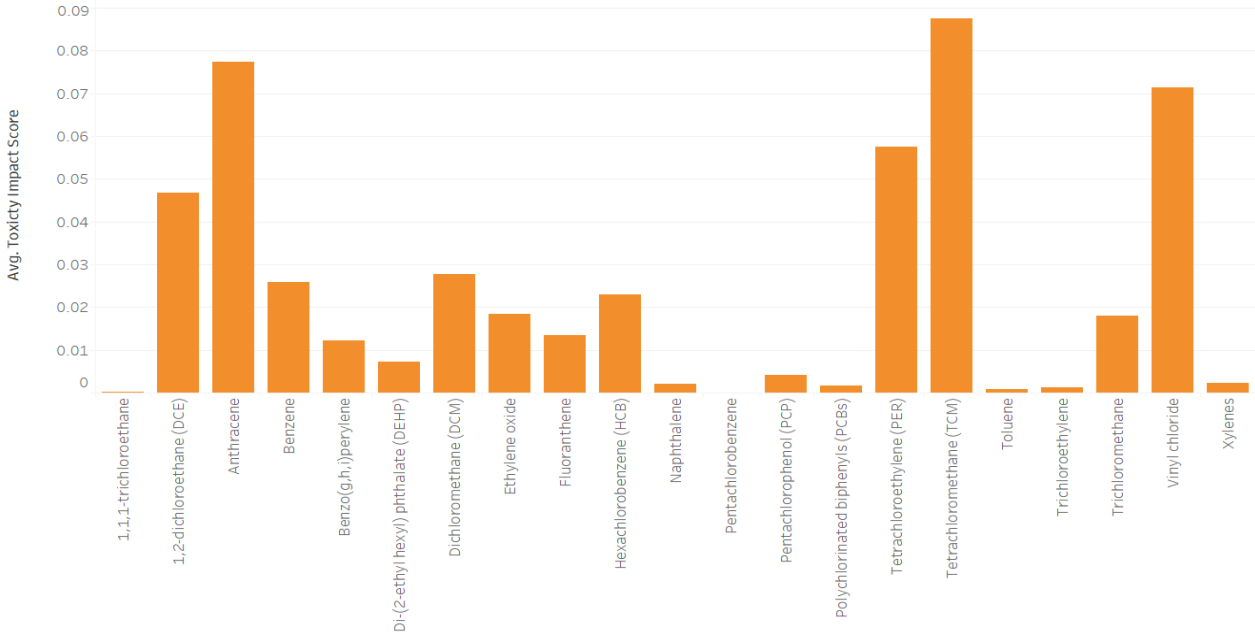


Figure 8: Average ecotoxicity of the non-metallic air pollutants reported to the E-PRTR between 2012-16

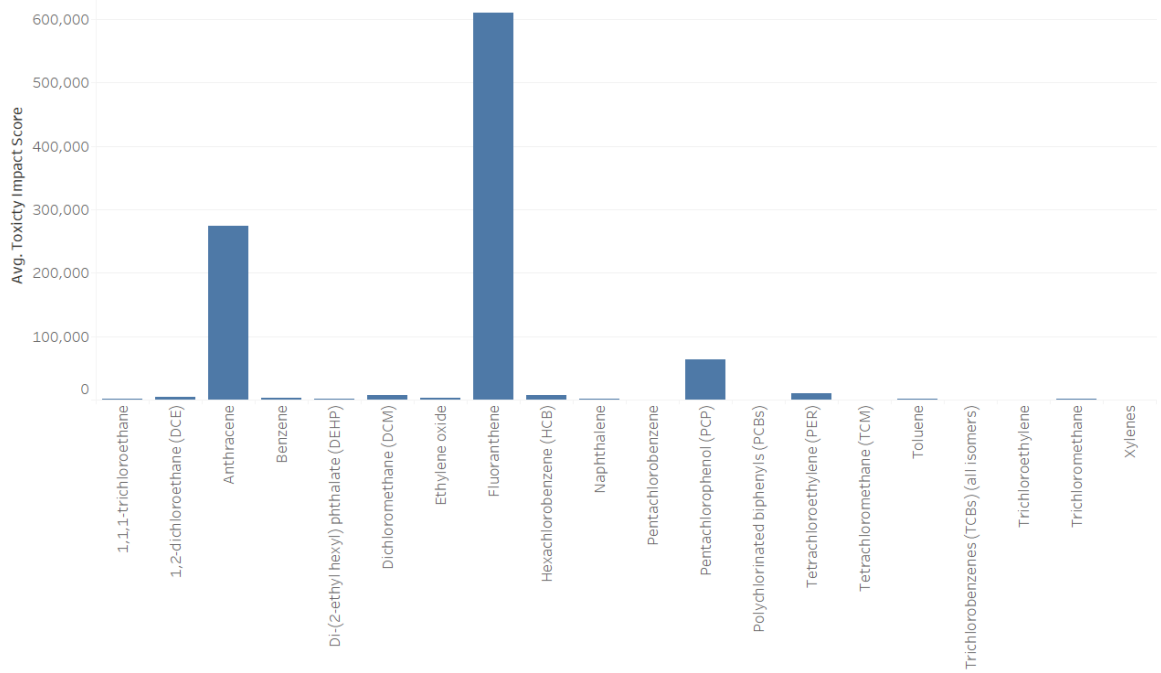


Table 4: Pollutants of interest based on toxicity score

Pollutant	Inclusion in future analysis?	Reasons
Anthracene	No	Despite quite high eco and human toxicity impact score of reported emissions, the number of countries and years over which it has been reported is low.
Fluoranthene	No	Despite having the highest eco toxicity impact score of the organic pollutants investigated, only one country reports this pollutant.
Dichloromethane (DCM)	Yes	The eco and human toxicity impact score of reported emissions is higher than benzene and reporting frequency is similar.
1,2-dichloroethane (DCE)	Yes	The eco toxicity impact score of emissions is higher than benzene and reporting frequency is similar.
Hexachlorobenzene (HCB)	No	While the eco toxicity impact score of emissions is higher than benzene the reporting frequency is low.
Tetrachloromethane (TCM)	No	While human toxicity impact score of emissions is the highest out of the organic pollutants investigated, the reporting frequency is relatively low.
Tetrachloroethylene (PER)	No	The eco and human toxicity impact score of reported emissions is higher than benzene however, very few countries report emissions.
Vinyl chloride	Yes	The human toxicity impact score of reported emissions is high and the reporting frequency is relatively high.

3.4 Preliminary suggestions for selecting pollutants

Based on an analysis of frequency of reporting in the E-PRTR and quantity of reported emissions, a first suggestion is to continue covering the pollutants assessed in earlier reports, i.e.

- ammonia (NH_3), nitrogen oxides (NOx), non-methane volatile organic compounds (NMVOCs), primary particulate matter ($PM_{2.5}$, PM_{10}) and sulphur oxides (SO_2),
- arsenic, cadmium, chromium, lead, mercury and nickel,
- benzene, dioxins/furans and polycyclic aromatic hydrocarbon.

A second suggestion is to additionally take into consideration methane and nitrous oxide, and carbon monoxide, copper, zinc, fluorine and chlorine.

An additional screening of ecotoxicity and human toxicity suggests vinyl chloride, dichloromethane and 1,2-dichloroethane might be worth including in the analysis as well.

The possibilities for assessing exposure for these pollutants and the availability of response functions and valuation data are further discussed in chapters 4 to **6**.

Some substances occur in different species or forms (e.g. organic, inorganic particle, gas …). In E-PRTR, generally, different species are accounted for, but they are not reported separately (e.g. Cr(III) vs. Cr(VI) are reported as “chromium and compounds”). The report in 2020 will describe how this is dealt with in the assessment.

Concerning country scope, most countries included in the E-PRTR can be covered by the analysis. The assessment will include the 27 EU Member States, the UK, Norway, Iceland and Switzerland. Serbia continues to be an exception, because of the lack of reported CO_2 data.

4 Dispersion and exposure modelling

Except for CO_2, impacts of all pollutants on human health and the environment are evaluated based on the Impact Pathway Approach. Depending on the complexity of the pollutant chemistry, on its dispersion and on the exposure route, different models may be mobilized. For the main air pollutants, such as particles and their precursors or O_3 and their precursors, inhalation is the only relevant exposure route for the human health impact. SOx, NOx and NH_3 have also environmental impacts through deposition of sulphur and nitrogen compounds responsible for eutrophication and acidification of water and terrestrial ecosystems (cf. chapter 5). Furthermore, ozone has harmful impacts on crops and forest, and can be responsible for loss in agricultural yields. The complexity of transformation and chemistry of these pollutants that involves non-linear processes requires the implementation of chemistry-transport models (CTMs) for the quantification of health and environmental impacts of those pollutants, for which inhalation and deposition are the main pathways for harmful impacts on health and ecosystems, respectively.

For metals and organic compounds, a multi-media approach is necessary, since not only inhalation is relevant for human exposure but also and predominantly for several pollutants, ingestion through consumption of foods and drinks. Modelling heavy metals and organic compounds should therefore include transfers in air, water and soil together with data related to ingestion of food and drinks to account for all exposure routes. Chemistry of these pollutants is, generally, less complex and simpler "passive" models can be used for the atmospheric air dispersion part. The exposure pathways were illustrated above in Figure 4 and cover inhalation and ingestion of contaminated agricultural produce, fish and water. Information on the approach followed previously, including transfer factors, was provided in Annex 3 of EEA (2014).

4.1 Main air pollutants

Modelling of main air pollutant dispersion and chemistry tracks pollutants in the atmosphere and follows their chemical reactions, enabling quantification of the atmospheric transport and transformation resulting from the release of primary emissions. An important consequence is that effects caused by secondary particulates or ozone are assigned to the primary pollutant (precursors) emissions from which they are formed (e.g.in the case of $PM_{2.5}$: SO_2 for sulphate aerosol, NO_x for nitrate aerosol and NH_3 for ammonium aerosol). The modelling also allows accounting for non-linear chemical interactions between air pollutants, for example the effects of NMVOC emissions on secondary organic aerosols, or the effects of NO_2 and NMVOC emissions on ground-level (tropospheric) ozone formation.

4.1.1 Situation in the previous assessment

The method used in the previous assessment (EEA, 2014), based on the Impact Pathway Approach, consisted in reducing emissions of a selected pollutant over each country individually, to estimate the associated reduction in concentrations and exposure to $PM_{2.5}$ and O_3 over Europe. Based on concentration-response functions, the associated reduction in health impacts and health costs were then calculated. By dividing the avoided health costs by the quantity of emissions reduced, a country-specific external cost per tonne of pollutant was estimated. It should be noticed that current response functions are developed to link outdoor concentrations to sanitary effects, even if the population actually spends part of its time indoor. Therefore, the method is limited to outdoor concentration responses to emission reductions.

CTM models are adapted to calculate air concentrations of pollutants over large regions such as Europe with spatial resolutions varying from 2km×2km to 50km×50km. Because the objective of the IPA here is to estimate a country-specific avoided external cost associated with emission reductions of NO_x, PPM, SO_2, NMVOC and NH_3, a full run with a CTM reducing independently each pollutant over each country would be heavily consuming in terms of computational time. For this reason, both the 2011 and 2014 EEA reports were based on the use of EMEP Source Receptor Matrices (SRMs) released each year by EMEP/MSC-W under the UNECE LRTAP Convention. These matrices are based on sensitivity simulations of the full EMEP/MSC-W Chemistry Transport Model. In a country-to-grid configuration, they give the change in various pollution levels in each receptor grid resulting from a change in anthropogenic emissions from

each individual country (or natural emitter region). Such matrices are generated by reducing emissions for each country (or region) of one or more precursors by a given percentage (15 % in this case). Over each country, the emission reductions of the five main air pollutant precursors (NOx, PPM, SO_2, NMVOC and NH_3) are modelled independently. But the reduction is not specific to the different anthropogenic sectors (industry, transport, domestic, agriculture, etc.).

In the 2011 report (EEA, 2011), the option of using the latest SRMs available at the time was considered in order to benefit from the recent improvement of the EMEP model. However, they were only available for the meteorological year 2006, which was considered outstanding (warmer than usual) and therefore not representative enough of the European average situation. A previous version of SRMs, based on a more representative year (1998), designed for the revision of the National Emission Ceilings Directive (NECD) and the UNECE Gothenburg Protocol was therefore preferred.

In the 2014 EEA report, the following changes in dispersion modelling were considered as being amongst the most important compared to 2011:

- The SRMs used were those updated for the revision of the EU Thematic Strategy on Air Pollution in 2013. A first benefit was that the EMEP model included then for the first time secondary organic aerosols modelling. Secondly, 5 meteorological years (2006 to 2010) were used (and averaged) to account for meteorological interannual variability.
- Sectoral adjustments were applied using results from a multi-model exercise: Eurodelta-II (Thunis et al., 2008) based on models with a 0.5° × 0.5° (around 50 km × 50 km) resolution and for the year 2000. The regular EMEP SRMs are only derived for each primary emitted pollutant and each country/region. It was therefore not possible to account for the fact that emissions of a given activity sector are, in reality, heterogeneously distributed over the country. The Eurodelta-II results allowed to compensate for this shortcoming, but only for France, Germany, Spain and the United Kingdom. In the Eurodelta-II project, for these countries, sector-specific potencies (referred to as "Correction Factor", in the 2014 EEA report) have been calculated to compare the impact on population exposure of homogeneous emission reductions with reductions over one specific sector. These country-and-sector specific potencies were applied to results from the SRMs to estimate a country-and-sector specific external cost for these 4 countries. For other European countries, in the absence of specific potencies, an average of the potencies from the 4 available countries was used. The use of these sectoral adjustments assumed that emissions from an E-PRTR facility can be associated with a specific sector (SNAP[10] sectors in this case, see Annex 1 for a description of these sectors). Because the E-PRTR sector categories are more aggregated than the SNAP nomenclature, it was chosen to attribute all emissions to the main activity sector reported in the E-PRTR data. A limitation of this approach is that it does not account for the fact that an industrial facility may have several activities that can be classified in different SNAP sectors. Using information on all concerned SNAP sectors for all industrial emission sources in E-PRTR would however be too time consuming.

4.1.2 New scientific knowledge and tools since the 2014 EEA report

<u>EMEP SRMs updating:</u>

EMEP SRMs are updated regularly and the results are publicly available.

In 2020, a renewal of the EEA assessment of externalities from industrial facilities would benefit from the recent developments in the EMEP SRMs[11] which include

- Improvement of secondary organic aerosol modelling that was at its infancy in the version used for the 2014 EEA report,

[10] Selected Nomenclature for Air Pollution, http://en.eustat.eus/documentos/elem_13173/definicion.html.
[11] https://emep.int/publ/reports/2018/EMEP_Status_Report_1_2018.pdf

- An improvement in model resolution was achieved in 2018, following the increase in spatial resolution of reported emissions to 0.1 x 0.1 degree in 2017. Given computational limitations, it was not possible to use directly the 0.1 degree emissions and the SRMs were produced on a resolution of 0.3 x 0.2 degrees (longitude x latitude) for the year 2016 (emission and meteorological data for 2016). These SRMs generally are produced every year for the year Y-2. EMEP/MSC-W are also working on a combination between SRMs at 0.3 × 0.2 degree resolution and the consistent base case to disaggregate SRM information to a 0.1 × 0.1 degree resolution domain. For the work in 2020, the most recent SRMs will be used, including disaggregated 0.1 × 0.1 degree resolution SRMs if available.

The shortcomings pointed out in previous reports with respect to the linearity of SRMs and the lack in accounting for the heterogeneity of distribution of industrial sources over a given country would, however, still hold. This issue is further discussed below.

Further development concerns the calculation of ozone impacts on crops and forests based on the PODy (Phytotoxic Ozone Dose above a threshold '*y*') indicator, rather than on the AOT40 (Accumulated Ozone exposure over a Threshold of 40 ppb) [12] indicator. However, SRMs for ozone fluxes are currently not produced by EMEP.

Air Control Toolbox (ACT) and the issue of non-linearity:

The design of an external cost "per tonne" relies fundamentally on a linearity assumption: the impact of x tonnes of pollutant from a facility is supposed to be equal to x times the impact of one tonne. Most of the integrated assessment models apply some linearity assumption, extrapolating from e.g. a 15% emission reduction sensitivity test to smaller or larger emission reductions. To account for non-linearity, a non-linear response function per tonne of emission would need to be designed. This is out of the scope here, but at least the errors due to these linearity assumptions are assessed. Until recently, only full-frame chemistry transport models could capture the complexity of non-linear atmospheric chemistry. But in 2018, the Copernicus Atmosphere Monitoring Service designed a new surrogate model that copes with this limitation: the Air Control Toolbox (ACT). The toolbox is designed to explore mitigation scenarios in the everyday Air Quality (AQ) forecast, hence the crucial need to account for uncertainties. Since it has now been in operation for more than one year, its results can also be used to assess the importance of non-linearities over the long term. This analysis is performed in section 0. Some shortcomings of ACT should be kept in mind: at present, only Europe-wide emission reductions for a given macro-activity sector can be assessed (and not country specific emission reductions), and pollutant emissions cannot be reduced independently for each chemical species (a homogeneous reduction of all industrial pollutant emissions is performed), although an evolution is planned by 2021.

The SHERPA model:

The findings of Eurodelta-II were previously used to estimate potencies accounting for variation in emission spatialization between emission sectors. There has not been any update of this multi-model exercise since 2010[13], so that the base year is now clearly outdated (2000) and the horizontal resolution much too coarse for current practices (0.5°, about 50 km).

[12] The sum of the differences between hourly ozone concentration and 40 ppb for each hour when the concentration exceeds 40 ppb during a relevant growing season, e.g. for forest and crops.

[13] http://www.eurosfaire.prd.fr/7pc/doc/1301907090_lbna24474enc_002.pdf?PHPSESSID=75bb8ddcaca78c97a1777f06549d5065

An interesting perspective consists in testing the SHERPA model being developed by the JRC[14]. SHERPA is also a surrogate model trained on a full chemistry-transport model. SHERPA grid to grid Source-Receptor Relationships (SRR) are constructed on the basis of a few full-CTM simulations (around 10). Two versions of the SHERPA tool exist. One is based on the CTM CHIMERE (Menut et al., 2014) run at 7km horizontal resolution for the meteorological year 2009, with emissions based on GAINS total emissions per country-pollutant-sector for 2010 and gridded with proxies from the MACC-TNO emission inventory from the year 2010 and specific national inventories for France and the UK (Thunis et al., 2016). This version is available online on the SHERPA webpage. Another version has been recently developed (Pisoni et al., 2019) based on the EMEP MSC-W model 4.9, for meteorological conditions from 2014, with a resolution of 0.1° and with emissions provided by JRC for the year 2014 (Trombetti et al., 2017).

Based on SHERPA grid to grid SRR, it is possible to evaluate the impact of NO_x, NMVOC, NH_3, SO_2 and primary particulate matter (PPM) emission reductions on $PM_{2.5}$ exposure. The selection of areas over which reductions can be applied varies from very local to country wide reductions. Another specificity of the SHERPA SRRs is the possibility to apply emission reductions over a particular sector (at SNAP level 1) instead of assuming homogeneous reductions over all sectors, as is the case in EMEP SRMs. SHERPA SRRs relate gridded emission changes to gridded concentration changes simulated by the CTM. This feature is not exempt from assumptions as the actual CTM sensitivity simulations underlying SHERPA do not explicitly isolate each activity sector. This means that when reproducing gridded concentration changes due to a reduction in an industrial sector, SHERPA estimates the response of CHIMERE to an averaged reduction applied over the mean vertical profile of all sectors (at the ground and at the height of the industrial source if included on the grid), and not to the specific height of the industrial sources targeted. This assumption has been addressed in the recent update of EMEP-SHERPA (Pisoni et al., 2019) which includes sectoral validation tests (albeit only for relative changes and not absolute deltas). Validation is however not complete as no tests of the ability of SHERPA to capture the model sensitivity country by country, and for both sectors and precursors, have been performed. Nevertheless, this particularity can be used to construct an updated potency accounting for sectoral adjustments for each of the EU-28 countries. This possibility is explored in section 0, discussing also the potential limitations.

Another interesting possible use of SHERPA concerns the characterisation of the health impacts from NO_2 exposure that were not included in the previous EEA (2014) assessment. Current recommendations from HRAPIE (2013) for quantifying health impacts associated to NO_2 exposure are based on 1) total NO_2 exposure for morbidity impacts and on 2) exposure to NO_2 values above 20 $\mu g.m^{-3}$ for assessing chronic mortality (see discussions in chapter 5).

The calculation of NO_2 exposure (i.e. the sum over all grids in the domain of the grid concentration multiplied by the grid population) requires high resolution modelling, since NO_2 is a local pollutant exhibiting high concentrations close to sources and a sharp decrease when moving away from them. A 50 km resolution for the SRMs, as in the previous assessment, or a 0.2×0.3 degree resolution as in the 2018 EMEP-SRMs are not enough to represent these spatial variations. A report recently published by VITO for the European Commission (VITO, 2017) focussed on NO_2 exposure assessment at a European scale. It highlighted the sensitivity of NO_2 population exposure to different modelling parameters. Model resolution was one important factor. The authors evaluated errors introduced by NO_2 concentrations smoothing over the model grid. For concentration-response functions with no NO_2 threshold, the errors introduced by smoothing NO_2 concentrations over a 7 km^2 grid (SHERPA's grid resolution) were evaluated to range from 5% to 17%. Even if errors on this order (5-17%) are not negligible, the SHERPA grid resolution starts to be acceptable for assessing NO_2 exposure.

These errors are evaluated to be much larger if the concentration-response function used for NO_2 health impacts includes a 20 $\mu g.m^{-3}$ threshold (VITO, 2017). Indeed, the smoothing effect will lead to reducing most of the NO_2 grid concentrations below this threshold. In that case, the spatial resolution of the above-mentioned tools (7, 10 or 25 km) is probably not satisfactory. Given the limited number of CTM simulations

[14] https://ec.europa.eu/jrc/en/news/sherpa-computational-model-better-air-quality-urban-areas

required to set-up SHERPA SRRs, a finer resolution could be reached over Europe if SHERPA was calibrated with higher resolution CTM simulations. The feasibility is therefore constrained by the possibility to run a full CTM at a resolution of e.g. 1km over Europe to estimate exposure associated with a 20 µg.m-3 NO_2 threshold.

The main limitations of the SHERPA model are the following:

- O3 concentrations are not calculated, it only covers impacts on NO2 and PM2.5 concentrations;
- There is a risk to overestimate the impact of industrial emissions with SHERPA. Indeed, in a full CTM industrial emissions are emitted at different heights for the different industrial sectors. This results in lower concentrations around the industrial sites compared to a modelling where all emissions were emitted at ground level. Even if SHERPA SRRs are constructed based on full CHIMERE runs, those runs are not specific to the Industrial sector, so that reductions of emissions for each targeted pollutant ignore the sectorial specification, and the responses smoothen out the effects of different sectors. Pisoni et al. (2019) estimated maximum errors associated with this sectorization around +/- 10%.
- It is based on a single meteorological year only.

4.1.3 Assessment of alternative options

Assessing the uncertainty associated with the linearity assumption

The Air Control Toolbox from the Copernicus Atmosphere Monitoring Service (CAMS) can be used to assess the uncertainties associated with the linearity assumption made when extrapolating the impact calculated with a 15% emission reduction. To achieve this, we compared the ozone, PM and NO_2 concentration changes associated to 100% emission reductions of the four main activity sectors (agriculture, industry, road transportation, and residential heating) to the concentration reduction reached with 15% emission reductions, linearly extrapolated to 100%. It appears that the non-linearities for PM are largest for the sectors agriculture and industry and for O_3 they are largest for the sectors industry and transport. Non-linear terms are largest when looking at individual days. But even for the annual mean there are locations in Europe, where the error can reach 9%, 8%, 8% and 2% for $PM_{2.5}$, PM_{10}, O_3 maxima, and NO_2, respectively.

To further illustrate this, we can compare the relative concentration change per tonne of emission over various countries estimated from sensitivity simulations based on either 100%, 15% or 1 tonne emissions reductions. The results are available in Table 5 for the impact of industrial emission changes on population-weighted $PM_{2.5}$ concentrations. If we take the example of France, the average population-weighted $PM_{2.5}$ concentration is 9.7µg.m^{-3}. Table 5 indicates that this exposure is reduced by 0.0462, 0.0419 and 0.0413 when computed from sensitivity simulations based on 100%, 15% or 1 tonne of industrial emissions. Non-linearities yield substantial differences when comparing the outcomes of 15% or 100% emission reductions, but the differences are much smaller when comparing the outcomes of 15% and 1 tonne reductions, and for all the countries considered, PPM emissions are significantly higher than 1 tonne. Therefore, we can conclude that the impact of non-linearities is marginal when using 15% sensitivity simulations extrapolated downwards to 1 tonne as it is proposed here.

On the contrary, we also show that the cost per tonne should not be extrapolated upwards to 100% emission reductions. In source allocation studies, when one assesses the overall burden of an activity sector in a given country, such a linearity assumption can yield 5 to 20% errors as illustrated in the last column of Table 5 that compares changes achieved with 1 tonne and 100% emission reductions.

Note that we presented here exclusively the results for industrial emissions. The ACT model is designed by activity sectors rather than precursor species. Therefore, in Table 5, we scaled the relative emission reduction (100% or 15%) in the industrial sector, by the fine primary PM emission reduction of that sector. But the resulting change in concentration also benefits from the concomitant reduction in other pollutant precursors. This limitation has limited impact on the key message in Table 5 that illustrates uncertainties

related to the linearity assumption of atmospheric response to industrial emissions changes. It would, however, yield a technical difficulty if one tried to account for this information about non-linearity in the cost expressed per tonne of pollutant. That is because we can only quantify here the non-linearity uncertainty for the whole industrial sector and all pollutant precursors together, rather than by individual pollutant precursor (e.g. per tonne of NOx, SOx, etc.).

Table 5: PM2.5 population-weighted concentration reduction averaged by country resulting from a reduction of 1 tonne of fine PM industrial emissions when computed from a reduction of industrial emissions by 100%, 15% or 1 ton, and corresponding primary emissions (tonnes). Results of the CAMS_ACT non-linear surrogate model.

EU28 countries	***Change per tonne (100% reduction)***	***Change per tonne (15% reduction)***	***Change per tonne (1 tonne reduction)***	***Ratio 1 tonne/100% reduction***	***Fine PM industrial emissions (tonnes)***
AT	0,0676	0,0601	0,0591	0,8747	4756
BE	0,0208	0,0181	0,0177	0,8523	6531
BG	0,0606	0,0505	0,0490	0,8087	8174
CY	0,0384	0,0362	0,0359	0,9327	1240
CZ	0,0811	0,0668	0,0649	0,8007	4034
DE	0,0504	0,0439	0,0429	0,8512	29910
DK	0,2099	0,1849	0,1858	0,8852	774
EE	0,0127	0,0113	0,0110	0,8684	8910
ES	0,0579	0,0542	0,0537	0,9290	18272
FI	0,1760	0,1576	0,1595	0,9063	4268
FR	0,0462	0,0419	0,0413	0,8923	33210
GB	0,0531	0,0446	0,0435	0,8187	16566
GR	0,0625	0,0559	0,0550	0,8794	15228
HR	0,1225	0,1082	0,1066	0,8702	2543
HU	0,0737	0,0632	0,0618	0,8387	5351
IE	0,0400	0,0354	0,0349	0,8733	3314
IT	0,0688	0,0640	0,0634	0,9210	21679
LT	0,0542	0,0471	0,0462	0,8521	3339
LU	0,0228	0,0199	0,0194	0,8516	678
LV	0,0484	0,0427	0,0420	0,8684	3898
MT	0,0112	0,0102	0,0101	0,9024	189
NL	0,0504	0,0440	0,0430	0,8531	3674
PL	0,0392	0,0330	0,0321	0,8181	36286
PT	0,0110	0,0107	0,0107	0,9668	22825
RO	0,0588	0,0491	0,0477	0,8105	18990
SE	0,0950	0,0865	0,0861	0,9064	11260
SI	0,0927	0,0833	0,0821	0,8855	847
SK	0,1085	0,0894	0,0872	0,8037	1907

Updating the sectoral potencies

Potency values (or “correction factors” in the 2014 EEA report and “efficiencies” in Thunis et al., 2018) reflect the normalized impact of an emission reduction over one sector compared to the normalized impact of a homogeneous reduction over all sectors. A potency higher than one implies that control measures will be more efficient for this targeted sector than calculated with the SRM. The potency values for the different sectors are interdependent (potency values cannot be positive for all sectors, there must be one or more sector with potencies below one). More details on the calculation of potencies are given in Annex 1.

To evaluate the feasibility of updating sectoral potencies, we tested both CHIMERE-SHERPA and EMEP-SHERPA with a similar methodology as in Eurodelta-II (only CHIMERE-SHERPA is publicly available, but the JRC made a version of EMEP-SHERPA available for the ETC/ATNI work). These versions of SHERPA are based on two different CTMs (CHIMERE and EMEP) run for different meteorological years (2009 and 2014), using different emission inventories (for the years 2010 and 2014) and different horizontal resolutions (7km and 10 km). Details on the methodology and on model configurations are given in Annex 1. Potencies were estimated for 6 countries, 4 sectors related to industry (SNAP 1, SNAP 3, SNAP 4 and SNAP 9, cf.

Table 6) and for NOx, PPM, SO2 and NMVOC.. Potencies are always calculated for the impact of a reduction in the primary pollutant on population-weighted PM2.5 concentrations. The main characteristics of these sectoral potencies derived with SHERPA are:

- a clear country sensitivity: for a given sector and pollutant, sectoral emission reductions can be more or less efficient in reducing exposure than are homogeneous emission reduction over the country;
- a significant spread in potency values (reaching a factor 2) is calculated for primary particulate matter emissions (PPM) in both directions. This means that for this pollutant, the impact on human health can be overestimated or underestimated by a factor of 2 when using external costs with no sector differentiation. For NOx, these errors are lower: within a range of 30%. NMVOC potencies are almost always below one, reaching 0.5 for the lowest case, meaning that this pollutant's impact can be overestimated up to a factor 2 when ignoring the sector differentiation. Finally, the pollutant for which sector differentiation is least important is SO2 (maximum errors estimated at +/-13%).
- In general, potency values are correlated with population-weighted emissions of the corresponding country. This is particularly the case for primary particles (PPM) that contribute directly to PM2.5 concentrations. The correlation is less important for SO2 or NOx that will be transported before contributing to PM2.5 concentrations after oxidation.

The results from CHIMERE-SHERPA and EMEP-SHERPA were compared with country specific potencies used in the previous assessment (Eurodelta-II factors from "#model 3", EEA (2014)). It can be noted that potencies for SNAP 9 (waste treatment and disposal) were not calculated in the previous assessment (EEA, 2014), and potencies for SNAP 4 (industrial processes without combustion) not systematically. This comparison is shown in Table 6 for NO_x, PPM and SO_2.

Table 6: Comparison between PPM, NOx and SOx potencies (relative to impact on PM2.5 concentrations) calculated with CHIMERE-SHERPA, EMEP-SHERPA and Eurodelta-II. Potencies higher than 1 are coloured in red. Values associated with too low emissions in a given SNAP sector or a too small contribution on concentrations are considered as not representatives (Low Rep: low representativity), see Annex 1 for more details.

PPM potency	SNAP1 Combustion in energy industries			SNAP3 Combustion in manufacturing industry			SNAP4 Production processes			SNAP9 Waste treatment		
Country	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta
France	Low Rep.	1,08	0,64	2,04	1,31	0,63	0,89	1,30	1,08	0,45	1,04	
Germany	1,24	1,13	0,51	1,22	1,11	0,55	1,11	1,16	1,38	1,12	0,93	
Spain	0,73	0,48	0,39	0,88	1,11	0,52	0,76	1,09	0,84	1,38	1,17	
United Kingdom	1,15	0,75	0,47	0,86	0,98	0,58	0,97	0,98	1,31	1,20	0,91	
Italy	0,98	0,76		0,95	1,45		0,87	1,34		0,57	1,12	
Netherlands	1,13			0,86			0,74			0,98		

NOx potency	SNAP1 Combustion in energy industries			SNAP3 Combustion in manufacturing industry			SNAP4 Production processes			SNAP9 Waste treatment		
Country	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta
France	1,03	0,94	0,91	1,01	1,02	0,87	Low Rep.	1,01		Low Rep.	Low Rep.	
Germany	1,06	1,01	0,80	1,05	1,03	0,84	1,00	0,54		Low Rep.	Low Rep.	
Spain	0,82	0,72	0,65	1,12	1,26	0,93	0,99	0,89		Low Rep.	Low Rep.	
United Kingdom	1,03	0,92	0,74	0,96	1,02	0,79	Low Rep.	0,88		Low Rep.	Low Rep.	
Italy	0,93	0,67		0,92	1,25		Low Rep.	Low Rep.		Low Rep.		
Netherlands	0,98	0,97		0,99	1,02		Low Rep.	0,96		Low Rep.		

SOx potency	SNAP1 Combustion in energy industries			SNAP3 Combustion in manufacturing industry			SNAP4 Production processes			SNAP9 Waste treatment		
Country	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta
France	0,99	0,98	0,74	1,02	1,04	1,06	0,99	0,83		Low Rep.	Low Rep.	
Germany	1,02	0,98	0,86	1,01	1,00	1,03	1,04	1,08		Low Rep.	Low Rep.	
Spain	0,93		1,01	1,08		1,03	0,88			Low Rep.		
United Kingdom	1,06	0,93	0,86	0,95	1,13	0,96	0,94	1,00		Low Rep.	Low Rep.	
Italy	0,96			0,96			1,10					
Netherlands	1,00			0,99			1,05					

Potency values reflect the relative impact of an emission reduction over one sector compared to a reduction of emissions over all sectors. The spatialization of the emissions and their magnitude influence these values.

Large differences are found between potencies derived from CHIMERE-SHERPA, EMEP-SHERPA and Eurodelta-II (EEA, 2014):

- For SNAP 1 (combustion in energy industries) and SNAP 3 (combustion in manufacturing industries), the Eurodelta-II model almost systematically estimates that emission reductions over these sectors are significantly less efficient than homogeneous emissions reductions over all sectors (potency lower than unity) for PPM and NOx. Besides, PPM and NOx potencies for the residential sector, dominated by residential heating, are higher than one due to the use of a proxy based on population density for the spatialization of this sector's emissions (not shown here, see Annex 1). CHIMERE-SHERPA and EMEP-SHERPA-based potencies for industrial sectors show a different behaviour, with significantly higher efficiency (potency higher than unity) in some countries. These differences can be partly explained by the proxy used to spatialize residential heating emissions in these models that include a parametrization to decrease PPM emissions per

inhabitant when the population density increases resulting in more emissions over rural areas where biomass burning is more frequent. In some countries this results in PPM potencies lower than 1 for SNAP2 and, as a mirror effect, higher than 1 for some industrial SNAPs. Differences may also come from the model resolution (50 km for Eurodelta-II and 7 and 10 km for CHIMERE-SHERPA and EMEP-SHERPA, respectively), model behaviour and also from the surrogate simplification related to the emission injection height.

- Potencies calculated with the 2 SHERPA versions also vary significatively between each other, partly showing contradictory results. These differences are attributed to different underlying emission inventories and CTM characteristics. The analysis in Annex 1 suggests that potencies are strongly driven by the spatialization of the emission inventory.
- For a given sector and pollutant, potencies evaluated with CHIMERE-SHERPA and CHIMERE-EMEP vary more across the different countries than in Eurodelta-II.

It should be noted that the use of CHIMERE-SHERPA also allows estimating potency factors for the impact of NO_x reductions on NO_2 exposure. These factors are given in Annex 1 for the 6 countries tested. Using these modulation factors would however only be of interest if EMEP SRMs at 0.1 degree of resolution became available. At present, they are only available at 0.2x0.3 degree of resolution, which makes them irrelevant for NO_2 exposure, therefore requiring the use of SHERPA, also for the country to grid sensitivity.

At this stage, we demonstrated the feasibility to use the CHIMERE-SHERPA and EMEP-SHERPA models to derive country-specific potencies that could be used to modulate the overall response provided by the EMEP SRMs, as done with Eurodelta-II potencies in the previous assessment (EEA, 2014). The analysis conducted in annex 1 shows that this response is very sensitive to the emission inventory used in the CTM underlying the SHERPA model and to the spatialization of non-industrial sectors. In some cases, industrial areas with high emission levels may dominate the overall population exposure values. But the assumptions related to injection height could also play a role.

The high variability between estimated potencies from different countries may be enhanced by the uncertainties in emission mapping for various countries. However, the spatial heterogeneity of potencies does reflect a reality in the relative location of industrial facilities and population centres.

Inclusion of additional pollutants

Pollutant screening in Chapter 3 suggested additional pollutants, such as vinyl chloride, dichloromethane, 1,2-dichloroethane, fluorine and chlorine, might be worth including in future EEA damage cost assessments. These pollutants are not likely to be included in chemistry-transport models (they are not in EMEP, CHIMERE). Exposure calculations would have to rely on other types of models or approaches.

4.2 Toxic metals and organics

4.2.1 Situation in the previous assessments

Any model involving assessment of the effects of persistent pollutants including toxic metals and organics has to account not only for uptake by inhalation, but also uptake through other routes, particularly ingestion of food and water. Exposure by routes other than inhalation can dominate the total intake of material by a factor of 10 or more (Rabl et al., 2014).

The multi-media modelling framework for quantifying pollutant uptake used for the EEA (2014, Annex 3) analysis shows the complexity required of the modelling (Figure 4). The approach taken involved the following steps:

1. Quantification of air concentrations using the Uniform World methodology described by Spadaro and Rabl (2004). A key parameter of the analysis is the bulk or total pollutant deposition velocity, which includes air removal by dry and wet mechanisms. For the organic pollutants, consideration was also given to chemical transformations. Parameters were varied by pollutant and country.
2. Assessment of environmental accumulation, transport and estimation of concentrations in soil and water compartments, leading either to direct human exposure (e.g. by drinking water) or to movement of pollutants into the food chain.
3. Quantification of uptake by plants and animals leading to human exposure via ingestion of foodstuffs.
4. Assessment of the passage of pollutants through the human body on the way to their ultimate environmental disposal, which may involve, for example, soil fixation or settling in waterbed sediment.

The movement of material through the chains just described and shown in Figure 4 was assessed using a Uniform World Model and transfer and other factors taken from the literature (sources are identified in the EEA report). Further information on response functions and valuation is provided in Sections 5.4 and **6.3**.

In future analysis, consideration should be given to new information on dietary intakes of different food and drink materials and on the cross-media transfer factors. The assumptions adopted in EEA (2014) around the use of European average values for pollutant contamination of food and drinks were intended as a reasonably pragmatic solution to the problem of accounting for variability, recognising that the long-range dispersion of pollutants from E-PRTR sites will lead to an averaging out of contamination. There may still be a few sites within Europe where emissions of specific toxic metals or organics can cause significant local contamination that is not captured by such modelling, but given the current state of environmental legislation for industry within the EU, the number of such sites should be few and diminishing.

More sophisticated models are available (see below). However, there are a number of benefits to using a simplified approach as adopted in EEA (2014):

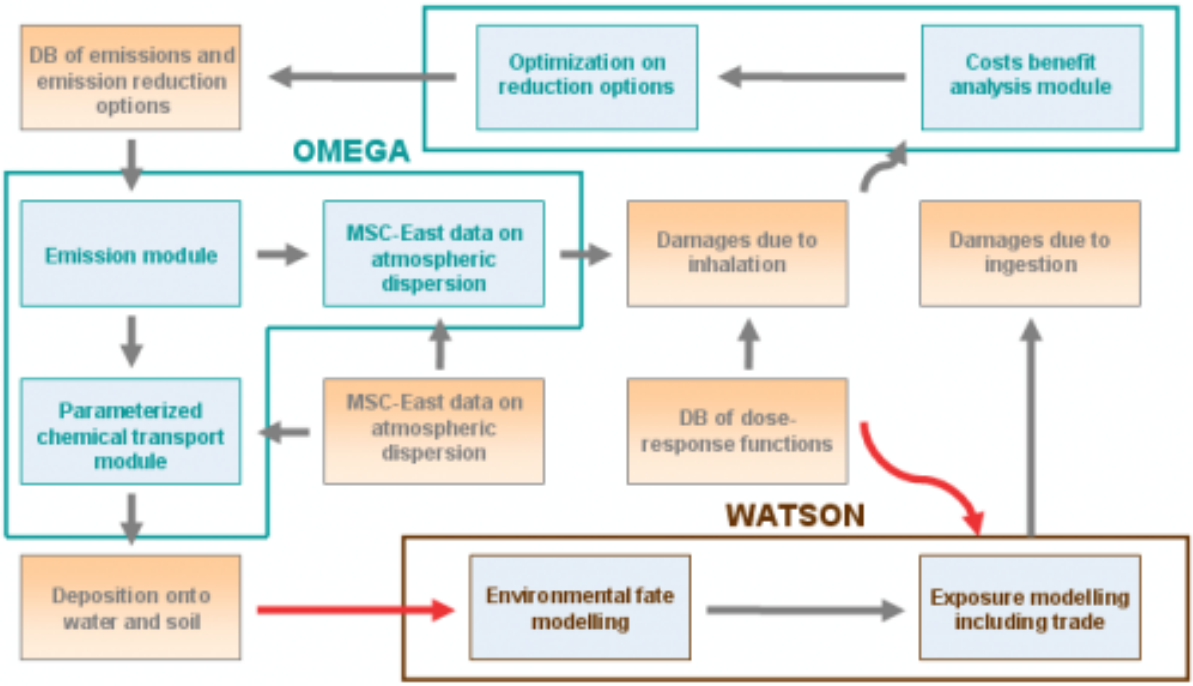
1. It is straightforward to check the logic of the modelling and the results generated.
2. The potential for significant error is limited.
3. The time required for generating damage estimates is limited.

4.2.2 New approaches

This section discusses two models, ESPREME[15] and Pangea.

The aim of the ESPREME study that ran from 2004 to 2007 was to perform damage assessment considering heavy metals releases to the environment and human health in the long term. The priority metals for ESPREME were mercury, cadmium, lead, nickel, arsenic and chromium. The ESPREME approach was adopted by the HEIMTSA and INTARESE studies under Framework Programme 6 of EC DG Research for these metals, (University of Stuttgart, 2011). The modelling framework of ESPREME is shown in Figure 9.

Figure 9: Dataflow of the ESPREME modelling framework.



The WATSON (integrated WATer and SOil environmental fate, exposure and impact assessment model of Noxious substances) Model is described in detail by Bachmann (2006). Air dispersion of toxic metals was assessed using the methods developed by MSC-East[16] under the UNECE Convention on Long Range Transboundary Air Pollution.

The overall modelling framework is similar to that used for the EEA (2014) report, following pollutants through environmental compartments to quantify human exposures and health impacts. Some of the individual models used within the ESPREME framework appear more sophisticated. Direct comparison of results for the exposure assessment specifically has not been possible here because the model is no longer supported, but a comparison of damage costs for some of the metals provides very similar results to those calculated for EEA (2014) per tonne emission. The need for the added complexity is thus questionable.

The developers of the Pangea model[17] describe it as "*a local to global, spatial multi-scale, multimedia environmental fate and transport, and multi-pathways population exposure framework for modelling chemical substances*". Currently available information concerns case studies in Asia (Wannaz et al., 2018a) dealing with home and personal care chemicals emitted to water, Australia (Wannaz et al., 2018b) dealing with benzene and formaldehyde emitted to air, and a global assessment (Wannaz et al., 2018c) of exposure

[15] http://www.integrated-assessment.eu/eu/indexa62b.html?q=resource_centre/integrated_assessment_heavy_metal_releases_europe.
[16] http://en.msceast.org
[17] http://www.pangea-model.org, containing a short description of the model and 3 key references.

to dioxins and furans, benzene and PAHs linked to waste to energy plants in France. A search for other published material using the model, particularly in Europe, has not identified further reports in the public domain, though it is understood to be used internally at Unilever. All three papers provide similar descriptions of the modelling framework, with further information on ecotoxicology provided in Wannaz et al. (2018a) and on human exposure in Wannaz et al. (2018b). The model brings together globally spatialised and consistently coupled data on meteorology, population, food production, etc., combines data with dispersion modelling and ends with estimates of environmental concentrations and population exposure 'at steady state'. The modelling system for the Pangea model is both sophisticated and referenced to other state-of-the-art materials. The modelling framework is flexible, allowing more detailed grids to be adopted in areas that contribute most to the total exposure linked to a specific source. Wannaz et al. acknowledge limitations of the Pangea model, for example the restriction to first-order fate and exposure processes and a focus of the ingestion pathways on production rather than consumption. Reflecting a long-recognised issue in Europe, the model demonstrates that restriction of the range of assessment to (e.g.) 100 km will generate only a small part of overall exposure for gaseous pollutants (Wannaz et al., 2018c). For benzene, the range required to reach 90% of the intake fraction for food and fish is 600 km, though this shrinks to 460km for dioxins and furans and 120 km for PAH (as BaP).

The assumptions used in Pangea behind the modelling of exposure though ingestion of food and drink are unclear, with respect to the exposure of the European population, bearing in mind extensive international trade in agricultural (etc.) goods.

The Pangea framework can deal with any pollutant but is likely to work best for unreactive pollutants (given the use of first order fate modelling). Published results address rather few pollutants, so actual performance for pollutants with complex pathways, such as mercury, cannot be judged based on the published materials.

Pangea is aligned with USEtox, which is based on model inter-comparison (SimpleBox, WATSON, EcoSense, CalTOX, UsesLCA). The link to EcoSense is useful from the perspective of transboundary transport of reactive air pollutants, though the EcoSense modelling may not reflect the current state of the art as represented by models like EMEP and CHIMERE.

4.3 Conclusions taking account of feedback from the expert consultation

Due to computation time constraints, it is still not feasible to use full model runs to estimate costs of air pollution from all individual industrial facilities per country for the ***major air pollutants***. The use of source-receptor matrices (SRMs) or surrogate models is still recommended.

For the estimation of the impact of NO_x, PPM, SO_2, NMVOC and NH_3 on $PM_{2.5}$ exposure, the use of the most recent and validated EMEP SRMs is preconised. These SRMs are commonly used and validated over Europe, and are updated every year. Concerning the O_3 impact on agriculture, AOT40 SRMs could still be used until PODy SRMs become available.

Errors associated with non-linear responses to small reductions in emissions are considered acceptable with regard to other uncertainties and errors. They can therefore be neglected when it comes to assessing a cost per tonne of pollutant. However, the uncertainty can reach 5 to 20% (depending on the country), when the linearity assumption is applied to an assessment of the impact of the overall industrial sector over a given country.

The desirability of deriving a unit damage cost (€/tonne of pollutant) specific to the industrial sector was identified in the EEA (2014) report. However, it is apparent that there is a lack of consensus on the best approach to dealing with this sector specificity. The results of the earlier EURODELTA II study are now outdated, so can no longer be regarded as state-of-the-art. CHIMERE-SHERPA and CHIMERE-EMEP based potencies calculated so far over 6 countries show large country variability and are very different from previous studies. Assessment carried out as part of this work finds that there are three options for proceeding:

1. Adopt average damage costs by country that do not take account of the specificity of the industrial sector. For pollutants with potency ratings close to 1 (i.e. close to the national average damage cost) from analysis so far using the SHERPA models (NOx and SOx), this position, though imperfect, may be a pragmatic solution to the problem. It is more questionable for PPM and NMVOCs.
2. Use the results of a specific model (SHERPA-EMEP or SHERPA-CHIMERE). As the SRMs used to estimate the impact of emission reductions on concentration are based on the EMEP model, the choice of SHERPA-EMEP would be more consistent.
3. Use results averaged across SHERPA-EMEP and SHERPA-CHIMERE.

At the present time, limited results are available from the SHERPA models. Further modelling is needed to better understand the way that the results for different sectors vary across Europe. In particular, countries distant from the centre of Europe (e.g. Greece, Portugal, Finland) should be added to the analysis prior to making a final decision on the most appropriate approach and source for sector adjustments. Inclusion of these countries will improve understanding of the extent and reasons for variation.

For the assessment of NO_2 health impacts, the use of the CHIMERE-SHERPA model with 7km×7km grid-to-grid SRRs is feasible, with estimated errors associated to the grid resolution below 20% (based on no-threshold indicators). There is an issue about the current resolution of the CHIMERE-SHERPA tool (7km) and its tendency to smoothen out NO_2 gradients, especially to estimate population exposure to NO_2 concentration levels above a threshold (such as 20 $\mu g.m^{-3}$ as in the response function currently recommended by the WHO/Europe). A more spatially resolved CTM-SHERPA would be useful to address this issue. Our suggestion is to use SHERPA (CHIMERE-SHERPA or EMEP-SHERPA) with its current resolution for the 2020 update for response functions without threshold. With the actual resolution of the available tools, it is not recommended to estimate NO_2 exposure to concentrations above 20$\mu g.m^{-3}$.

Table 7 sums up our recommendations based on our screening and assessment of new tools and methods. For example, the calculation of external costs for NO_x emissions will require the estimation of $PM_{2.5}$, NO_2 and SOMO35 exposure for health impacts and AOT40 for O_3 impacts on crops.

Table 7: Recommended exposure routes

Health or environmental effect	Targeted pollutant					Health indicator	Recommended SRM	Use of industrial potency
	NOx	PPM	SO_2	NMVOC	NH_3			
$PM_{2.5}$ impact on health	×	×	×	×	×	$PM_{2.5}$ exposure per country (for all EU-28 countries)	Updated EMEP SRM	To be decided
NO_2 impact on health	×					NO_2 Exposure per country (no threshold)	SHERPA SRM	Calculation by SNAP sector
O_3 impact on health	×			×		SOMO35 exposure	Updated EMEP SRM	No
O_3 impact on vegetation	×			×		AOT40	Updated EMEP SRM	No
Eutrophication	×			×	×	Deposition of nitrogen species	Updated EMEP SRM	No

Concerning ***metals and other pollutants that can cause damage through a variety of pathways covering ingestion as well as inhalation***, there is variation in modelling frameworks, between simplified tools as used in EEA (2014) and more complex systems such as Pangea (Wannaz et al., 2018a, b, c). The consistency of the Pangea framework with existing European tools (EMEP, CHIMERE, etc.) for air pollution assessment is not known in detail. Consistency with other tools, however, seems good (e.g. EUSES, given the links with USEtox). There is a balance to be struck between complexity, transparency and flexibility. Without systematic testing of the different tools it is not clear which are best suited to the current application.

Perceived increase in accuracy of the exposure assessment for toxic metals and specific organic pollutants of relevance to the work under discussion needs to be seen in the wider context of the final estimates of damage per unit emission: discussion below highlights greater uncertainties at the stage of defining which impacts should be quantified than seem likely to originate from the exposure assessment. We therefore suggest that exposure methods do not need to be updated compared to the 2014 assessment, and that it is more appropriate to spend time refining the impact assessment. This position may not hold if the list of pollutants requiring quantification was to be extended.

5 Quantification of impacts

5.1 Situation in the previous assessment

There have been three sets of air pollution damage costs produced for the European Commission and its Agencies since the early 2000s:

- The first set is included in the BREF-13 on Economics and Cross Media Effects published in 2006 (European Commission, 2006).
- The second set was included in the EEA report 'Revealing the costs of air pollution from industrial facilities in Europe', published in 2011 (EEA, 2011).
- The third set was included in the 2014 EEA report (EEA, 2014).

The older results are now outdated and should not be used, following developments in dispersion modelling, health (etc.) impact assessment and valuation. The overall structure of analysis, however, follows the Impact Pathway Approach (IPA, see section 2) and has not been changed throughout the series of studies.

Consideration was given to quantification of water pollutant damages for the EEA studies, but data limitations, and the strong specificity of damage to the site of emission (i.e. water body accepting the discharge), prevented analysis.

EEA (2014) provides costs for emissions of:

- Major air pollutants: NH_3, NOx, $PM_{2.5}$ / PM_{10}, SO_2 and VOCs, accounting for exposure to the primary pollutant (as emitted to air) and to secondary pollutants such as fine particles and ozone that are formed subsequently through chemical reactions in the atmosphere
- Metals: arsenic, cadmium, chromium-VI, lead, mercury and nickel
- Organics: 1,3 butadiene, benzene, dioxins and furans, formaldehyde, PAHs (polycyclic aromatic hydrocarbons) and diesel exhaust
- Greenhouse gases

The unit damage costs for emissions of the major air pollutants are provided at the national level and hence are country specific. In this context, 'the national level' concerns the impact of emission of one tonne of pollutant from a particular country, wherever the impacts occur (i.e. specifically including transboundary impacts). To do otherwise would work against the polluter pays principle.

The 2014 damage costs provide estimates of damage per tonne, averaged across all sources of each pollutant in a country (i.e. industry, transport, domestic, agriculture, etc., noting that Annex 4 of the EEA report describes methods for adjusting values so that they are better representative of industrial emissions but only for 4 European countries[18]). The data sources used were as follows:

- Pollutant dispersion and chemistry: EMEP model transfer matrices, as used in the GAINS model to inform development of EU's Thematic Strategy on Air Pollution.
- Population data: Eurostat
- Response functions: HRAPIE (Health risks of air pollution in Europe) study carried out by WHO-Europe (WHO, 2013a) on behalf of the European Commission and involving numerous experts from European academic and health institutes, and also some North American experts. HRAPIE involved extensive review and discussion of the epidemiological literature available at the time. Results also included analysis of damage of ozone to crops using concentration (AOT40) based response relationships and of acidification to building materials. Analysis here drew on earlier

[18] The scope of work for the EEA (2014) assessment did not extend to new dispersion modelling. The sector specific assessment was therefore based on the work of the EURODELTA2 study where only 4 countries were considered specifically. Factors for other countries were taken as the average of these estimates.

work, though used accepted European response functions. Original work on these receptors was not considered necessary given that the overall damage costs are dominated by health impacts. Response functions for damage to crops and building materials were taken from ExternE (2005), dependent to a large extent on work that has been carried out in relation to the UNECE Convention on Long Range Transboundary Air Pollution.

- Valuation data: As adopted for cost-benefit analysis of the Thematic Strategy on Air Pollution (Holland, 2014) with average EU values applied in all Member States.
- PM_{10} damage was directly related to $PM_{2.5}$ using a factor of 0.65, based on the rough assumption that 65% of PM_{10} is composed of the finer fraction and that health impacts are attributable to the finer fraction.[19] In reality, the fractional share of PMs is of course source-dependent. Data are now available providing sector specific estimates of the fraction of $PM_{2.5}$ in the PM_{10} fraction (DEFRA, 2019[20]).

The damage costs for each pollutant cover impacts on human health from exposure to fine particles (both primary and secondary species) and ozone. For NH_3, NOx, fine particles, SO_2 and VOCs, they also cover impacts on materials and crops where appropriate.

Analysis of organics also considered uptake via inhalation and ingestion as appropriate to the pollutants and impact pathways considered. For ingestion, a similar approach was adopted to the metals where it was assumed that European-average transfers between media would provide the most robust description of exposure.

Greenhouse gases were costed using information on marginal control costs rather than damage costs. Reasons for this choice are discussed below.

Potentially significant limitations of the 2014 analysis concern the treatment of NOx and VOC emissions. For NOx, HRAPIE recommended functions to account for effects of exposure to NO_2, though these effects were not quantified in the damage cost estimation given concerns raised in discussions between the modellers and the epidemiologists about the ability of the pollutant modelling to provide an estimate of exposure consistent with that used to characterise the exposure-response functions from the epidemiological literature. Specifically, epidemiologists felt that exposure was underestimated by the models. The problem may be a function of the scale of the modelling, with the health analysis considering individuals living in areas with high levels of exposure (e.g. close to busy roads) whilst the modelling work averages out exposure at a much coarser resolution (cf. also discussion in chapter 4). The European Commission has funded further work to provide an improved modelling framework in this area, though this has yet to be applied for damage cost estimation (VITO, 2017; cf. also chapter 4). A further concern on this point is that the HRAPIE recommendations on NO_2 may now be outdated, given the publication of a significant quantity of new research on the pollutant since HRAPIE was finalised (as reviewed by COMEAP, 2018). Principal findings of the COMEAP assessment were that there was no evidence for a threshold[21] for effects associated with NO_2, but that response is likely to be weaker than indicated by HRAPIE per unit of exposure. Some of the COMEAP authors questioned a role for NO_2 per se, rather than potential correlated variables, though there was agreement that the functions adopted for PM did not describe the full burden of polluting activities.

A limitation of the analysis of VOCs arises because the assessment includes only impacts associated with exposure to ozone and secondary organic aerosols. Other impacts of VOCs, such as direct health effects of

[19] UBA (2019) makes the assumption that PM_{10} consists to 70% of $PM_{2.5}$ and to 30% of PM_{coarse}. Coarse particles have an aerodynamic diameter ranging from 2.5 to 10µm ($PM_{10-2.5}$).

[20] https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/770576/air-quality-damage-cost-guidance.pdf, page 12.

[21] The HRAPIE report selected a value of 20 ug/m^3 for chronic mortality effects, only because of the lack of epidemiological evidence below this concentration level. The 20 ug/m^3 level in HRAPIE is therefore really a cut-off level, and not a threshold value below which the health effects are zero.

exposure, were not included in most cases (leaving aside the separate analysis of benzene, formaldehyde and dioxins and furans). A more disaggregated analysis of VOCs by species would be possible if needed: this could take into account any direct health and ecological impacts of a substance to the extent that response functions are available, and variation in the potential of individual VOCs for photochemical oxidant creation, organic aerosol formation potential, etc. However, currently produced available receptor matrices do not include individual VOCs, and so population or ecosystems exposure would have to be assessed by other approaches[22].

The analysis of impacts of the selected metals considered pathways via the ingestion of food, milk and water as well as air inhalation for pollutants. Analysis considered a range of impacts (IQ loss for lead and mercury, cancers for nickel, cadmium, etc.) that are often used in risk assessment analysis, for example under the REACH Regulation. Limitation to the effects most often studied, however, may lead to significant underestimation of associated impacts. Nedellec and Rabl (2016a, b) performed a far more extensive analysis and generated much larger damage costs than earlier studies. Given the presence of national and international markets for food, and the widespread dispersion of pollutants from industrial sites, analysis of impacts via ingestion was performed using averaged modelling across the European region. Damage costs were country specific only to the extent that they concerned uptake via inhalation. A difference in the Nedellec and Rabl (2016a, b) analysis compared to the previous EEA (2014) work is the fact the Nedellec and Rabl study attributed the implied loss of life years lost per death to the birth cohort, rather than to the population alive today, with the latter loss being lower.

5.2 Major regional air pollutants

The major regional air pollutants considered are again NH_3, NOx, SO_2, VOCs, $PM_{2.5}$, and PM_{10}.

Since the earlier analysis was completed, a large body of new epidemiological and other health literature has been published. Notable examples include the following:

- RCP/RCPCH (2016): This study reviewed knowledge of air pollution and considered particularly evidence for impacts throughout the life course from conception to death. Standard analyses of air pollution effects focus mainly on the elderly, as this group is most likely to be admitted to hospital for respiratory or cardiovascular disease, or die. The RCP/RCPCH study highlighted the potential for far-reaching impacts on the young as well as the old[23]. However, provision of updated sets of response functions was beyond the remit of the research.
- Global Burden of Disease (GBD), (Institute for Health Metrics and Evaluation – IHME; Cohen et al., 2017): Air pollution impacts have been demonstrated through the GBD work to provide substantial damage to health around the world. Quantification of health impacts relies on the Integrated Exposure-Response model (IER).
- The Global Exposure Mortality Model (GEMM) Estimates (e.g. Burnett et al., 2018) suggests even larger health effects from $PM_{2.5}$ and health benefits from reductions in $PM_{2.5}$, especially at higher concentrations.

The GBD study considers the burden of disease generally, rather than specific to air pollution. In quantifying impacts at this broad scale consideration has been given to the selection of exposure response functions, focused especially on mortality linked to exposure to fine particles, as this has been found to be the major source of air pollution damage over the past 30 years. Historically, research has largely focused

[22] CTMs are capable of modelling individual VOCs but, as explained in chapter 4, full CTM model runs for quantifying costs of air pollution of all individual E-PRTR facilities are not feasible.

[23] The focus of the study covers effects of prenatal and childhood exposure to air pollution on susceptibility to chronic disease over the life course, including respiratory disease, cardiovascular disease, systemic effects such as diabetes, obesity, central nervous disease and cancer, as well as effects that maternal exposure to air pollution has on the developing foetus, such as miscarriage, stillbirth, premature delivery and low birth weight (RCP/RCPCH, 2016).

on assessment of populations in Europe and North America, leaving questions about the applicability of response functions to other parts of the world. These questions are specifically focused on two issues:

- Extrapolation of response functions to regions where pollutant levels are much higher than in Europe and North America
- Extrapolation to regions where health status is different to Europe and North America

More recently, there is increasing evidence from other areas around the globe. European analysis (based on WHO, 2013a) has used a linear function for all-cause mortality linked to $PM_{2.5}$ exposure. In contrast, GBD uses a series of non-linear, age specific, cause-specific functions to account for the issues just raised. There is, however, no evidence to suggest that this approach provides a result that is better suited for European assessment than the use of a linear function for all-cause mortality, for which national health statistics are more comparable across Europe and have greater precision (WHO, 2013a). Furthermore, several assumptions[24] are made in deriving the GBD functions that are not required when functions can be extracted directly from the epidemiological literature. Unlike the IER model, the GEMM estimates of mortality from exposure to $PM_{2.5}$ are based on cohort studies only of outdoor air pollution but that cover the global exposure range (including Chinese cohorts). Furthermore, baseline mortality rates refer to noncommunicable diseases (NCDs) – close to all-cause mortality in Europe - whereas the GBD quantifies mortality for 5 specific causes of death. GEMM hazard ratio predictions display a supra-linear association over lower exposures and a near-linear association at higher concentrations. Given the availability of Europe specific epidemiological studies there is no need for applying the standardised GBD risk functions that are used, among other reasons, for assessing the burdens based on the same set of response functions.

Recent revision of damage factors for the UK Department for Environment, Food and Rural Affairs (Defra) (Ricardo, 2019) includes a large number of health (mortality, morbidity) and other endpoints (underlying response functions are presented in Public Health England (PHE, 2018) in charge of a review of literature and data), many of which, such as effects on diabetes and stroke, not being quantified in the European work, or not being quantified for the same pollutants. The following list shows the quantified effects, with those shown in bold included in the EEA (2014) assessment.

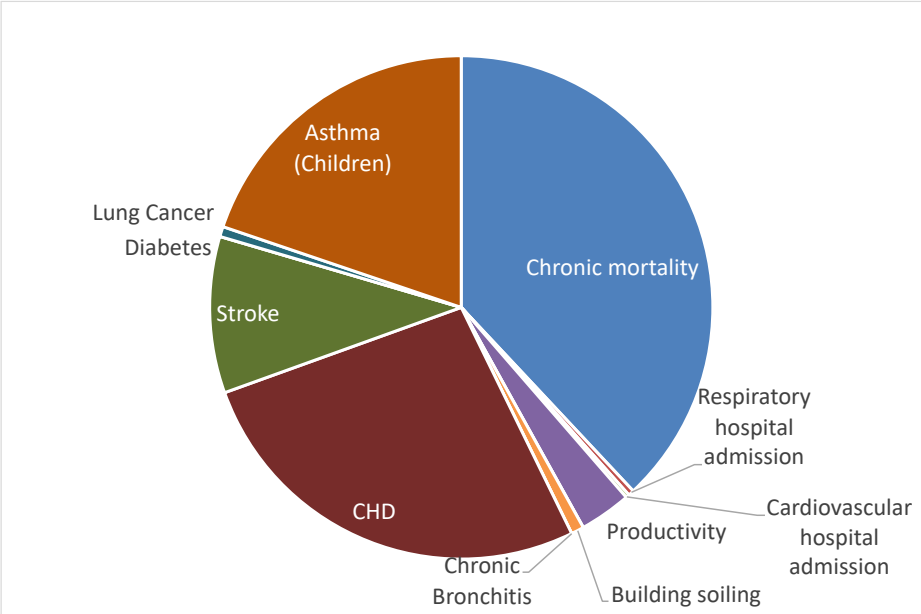
- NO_2, **$PM_{2.5}$: Chronic mortality**
- **O_3,** SO_2: **Deaths brought forward (from acute exposure)**
- NO_2, $PM_{2.5}$: Lung cancer morbidity
- **PM_{10}: Chronic Bronchitis**
- $PM_{2.5}$: Stroke
- $PM_{2.5}$: Coronary Heart Disease
- NO_2, **$PM_{2.5}$: Asthma in children**
- NO_2: Asthma in adults
- NO_2, **O_3, PM_{10}**, SO_2: **Respiratory hospital admissions**
- **O_3, PM_{10}: Cardiovascular hospital admission**
- NO_2, $PM_{2.5}$: Diabetes
- O_3, **$PM_{2.5}$, PM_{10}: Productivity**
- NH_3: Ecosystems
- NO_2: Ecosystems
- O_3: Ecosystems **(partial included in EEA (2014) via effects on crops)**
- SO_2: Ecosystems
- PM_{10}: Building soiling

[24] Such as the assumption of an independence of health impacts from the temporal pattern of exposure, the independence of different types of exposure, implying that there is no interaction between different types of exposure (Lim et al., 2012; Burnett et al., 2014).

- O_3, **SO_2: Material damage**

The Defra analysis is divided into three sensitivity cases, low medium and high. The number of impacts included in the sensitivity cases is (naturally) greatest for the 'high sensitivity' case[25]. The inclusion of the additional endpoints demonstrates growth in the epidemiological literature in recent years. However, for a number of effects there is as yet little mechanistic understanding of the link between air pollution and disease. Given that functions may be based on a very limited body of evidence the robustness of analysis for some endpoints included by Defra especially in the high sensitivity case is questionable. This applies particularly to a number of additional endpoints (e.g. diabetes) that make a significant contribution to total damage estimates (from review of material provided in the PHE (2018), report that provides much information on the novel endpoints introduced to the Defra work). Analysis using the assumptions of the HRAPIE study (WHO, 2013b) indicates that for $PM_{2.5}$, chronic mortality accounts for around 70% of the total impact (e.g. Holland, 2014). However, under the Central assumptions of Ricardo (2019), this falls to around 40%, with significant inputs also from childhood asthma, stroke and chronic heart disease (Figure 10). The Ricardo analysis for Defra demonstrates that further review of response functions is desirable. It is also noted that there is interest in further endpoints than those listed above, including dementia and premature birth / low birth weight.

Figure 10: Contribution of different health impacts and building soiling to total damage under the Central assumptions adopted by Ricardo (2019).



[25] In the 3 sensitivity cases not only does the range of impacts covered vary, but also the relative risk estimates of each impact. Therefore, the exact contribution of the additional impacts to the overall damage cost estimate cannot easily be factored out of the Ricardo report.

From the Ricardo analysis, the following 'new' endpoints provide significant contributions to total damage at least under the 'high sensitivity' case:

- NO_2: Diabetes and asthma
- O_3: Productivity
- $PM_{2.5}$: Coronary heart disease, productivity and stroke

The analysis of productivity impacts included in the Ricardo assessment is worthy of further consideration, given that it takes a more comprehensive approach than the assessment of work loss days that went into the previous damage costs for the EEA. The other impacts, however, may be too uncertain for inclusion in the best estimates at the present time. It is possible to apply them to derive an upper bound estimate of damage costs, useful not only in the context of the current workplan focussed on the impact of industrial emissions, but also for assessment of which impacts outside the current 'core' set may be most important and should be prioritised for further assessment.

The Defra (2019) and EEA (2014) results for the UK are compared in the following table.

Table 8: Comparison of UK and EEA damage costs for major air pollutants using results from Defra (2019) and EEA (2014). Units: EUR/tonne of pollutant, 2005 prices.[26]

	Defra (2019)	**EEA (2014)**
NH_3	€6,000	€9,500
NOx	€6,200	€3,600
$PM_{2.5}$	€105,000	€38,393
SO_2	€6,300	€14,425
VOCs	€102	€1,450

There are significant differences between both sets of data. The main reasons are:

1. Inclusion of domestic and non-domestic impacts in the EEA dataset (Defra (2019) includes only UK to UK impacts). It would be possible in future EEA work to quantify the domestic and non-domestic damage attributed to emissions in each country as it simply requires model outputs to be specified to include a breakdown by country: indeed, the transfer matrices used previously include the necessary level of detail to do this.
2. Quantification of a larger number of health impacts by Defra (2019) for fine particles.
3. Variation in response functions used.
4. Differences in valuation data.

[26] Converted to 2016 prices the values are:

	Defra (2019)	**EEA (2014)**
NH3	€7,700	€12,250
NOx	€8,000	€4,600
PM2.5	€135,400	€49,500
SO2	€8,100	€18,600
VOCs	€132	€1,870

Economic data in this working paper are converted to € price base 2016. Conversion between different currencies relies on annual average exchange rates (source: https://www.ofx.com/en-au/forex-news/historical-exchange-rates/yearly-average-rates/). Conversion between different years is based on the harmonised index of consumer prices (HICP) published by Eurostat (HICP (2015 = 100) - annual data (average index and rate of change) [prc_hicp_aind]). No regional adjustment was made to account for different income of countries.

Denmark, in its Air Quality Monitoring Programme (DCE, 2018) included in its assessment of externalities from air pollution several health endpoints not included in the previous EEA (2014) assessment: congestive heart failure, lung cancer, bronchodilator use (adults, children), cough (adults, children), acute premature death from SO_2. Response functions are taken from the review by Bønløkke et al. (2011). Based on the EVA modelling system, total external costs from air pollution in Denmark are presented for domestic and non-domestic impacts[27]. In a study updating the monetary unit costs for health impacts for Denmark's Ministries (Andersen et al., 2019) the list of health impacts considered is similar to the effects considered in EEA (2014).

Some recent publications apply response functions that indicate a potential increase of mortality linked to fine particulate matter pollution compared to previously estimated numbers. Lelieveld et al. (2019) use cause-specific mortality functions based on Burnett et al. (2018, cf. above). Their mortality estimates for Europe more than double relative to the Global Burden of Disease study. Relative to assessments made for the European Commission (e.g. assessment of the Thematic Strategy on Air Pollution, Holland, 2014) their results indicate a more modest 36% increase for Europe as a whole, but nearly 90% for the EU28 (this disparity does not appear to be attributable to differences in the area modelled). The GEMM functions (Burnett et al., 2018), actually, show supra-linear behaviour at low concentrations compared to straight line behaviour in HRAPIE response functions. Furthermore, the work with the IER and GEMM models use counterfactual concentrations of 2.4 ug/m^3 while the level is 0 for HRAPIE. Pascal et al. (2016), using a different approach based on log-linear functions for all-cause mortality, report figures for France that equate to a 24% increase for France, again relative to the analysis carried out in the context of the Thematic Strategy on Air Pollution. Applying the linear HRAPIE and the log-linear French dose-response functions to the same set of exposure data in France results in limited differences in estimated mortality numbers. For an assumed aggregated annual average exposure of 30 µg/m^3 the HRAPIE response functions leads to 12% higher mortality numbers, for an exposure of 12 µg/m^3 the difference is about 6%.

In late 2018 and early 2019 the German Environment Agency (Umweltbundesamt, UBA) published an update of its Convention on methods for determining environmental costs, including cost assessments for emissions of air pollutants and greenhouse gases (UBA, 2018 & 2019). As far as health impacts are concerned, they follow an approach comparable to that applied in the 2014 EEA assessment: modelling of exposure based on the impact pathway approach (here based on the EcoSenseWeb model developed in the NEEDS project) and quantification of health impacts based on HRAPIE (WHO, 2013a) and Holland (2014). Impacts are assessed for particulate matter ($PM_{2.5}$, PM_{10}, PM_{coarse}), NOx, SO_2, NMVOC, NH_3. Impacts on crops from ozone are based on dose-response relationships given in Mills et al. (2007), i.e. quantification of impacts using the indicator AOT40 and not yet the ozone flux approach. Results are presented per precursor pollutant (NOx, SO_2, NMVOC, NH_3). For some crops not covered, costs were estimated based on updated NEEDS data. The latter approach was also applied to an estimation of impacts on materials/buildings from SO_2 and losses in biodiversity. Results of the estimated average costs per tonne of pollutant are presented in chapter **6**. UBA (2019) recommends the use of differentiated, location specific cost estimates, especially for particulate matter emissions (cf. chapter **6**).

The Air Pollutant Marginal Damage Values Guidebook for Ireland 2015 (EnvEcon, 2015), provides damage per tonne of pollutant estimates for the major air pollutants that are differentiated between urban and rural areas as follows: Ireland all, rural, urban large (Dublin), urban medium (Pop ≥ 15,000), urban small (Pop 10,000 - 15,000), small towns (Pop < 10,000).

An Air Pollution Damage Cost Model (IHKU) has also been developed for Finland[28], with results being described by Savolahti et al. (2018). The model regionalised damage costs for Finland in two ways. For

[27] Total air pollution in Denmark, foreign contribution to Denmark, Denmark's contribution to Denmark, Denmark's contribution to Europe incl. Denmark and Denmark's contribution to Europe excl. Denmark.

[28] https://www.syke.fi/en-US/Research__Development/Research_and_development_projects/Projects/Air_Pollution_Damage_Cost_Model_for_Finland_IHKU.

industrial sources with high stacks, the country was split into Southern, Northern, and all Finland, whilst for low level sources the split distinguished urban and rural areas, as well as 'all of Finland'. Special interest was given to wood stoves and sauna stoves. The approach and values taken are broadly similar to those used in EEA (2014) though there are some differences.

The consultancy CE Delft, in its "Environmental Prices Handbook EU28 version" (CE Delft, 2018), has updated its assessment of environmental shadow prices for air pollutants and greenhouse gases. Costs for air pollutants are assessed using the IPA approach, with values being based on the NEEDS (2008a, b, c) study. Costs for CO2 are, as in the earlier study (CE Delft, 2010), based on the abatement-cost method.

Several studies commissioned by the European Commission dealing with impact quantification and valuation have recently been published or are currently ongoing.

- DG ENV commissioned the study "Mapping objectives in the field of environmental taxation and budgetary reform: Internalisation of environmental external costs". It should be finished by the end of 2020.
- There is an ongoing study commissioned by DG ENER on energy costs, taxes, subsidies and external costs (Trinomics, Enerdata), which aims to address the monetisation of external costs for energy products used in energy, transport and agricultural sectors. It should be published in June 2020.
- A Transport externality study is already published (https://ec.europa.eu/transport/themes/sustainable-transport/internalisation-transport-external-costs_en), including the "Handbook on the external costs of Transport", Version 2019 (EC, 2019).

The ongoing DG ENV study includes a task on external cost indicators aimed at identification and quantification of external cost indicators and associated unit values for impact endpoints of pollutants. The plan is to include in this review also cost estimates from national approaches. In the context of this project it was mentioned that the EC has an interest in the methodologies of the studies commissioned by the different DGs being aligned to avoid inconsistencies. Results of communication with the authors is presented in chapter 6.

The DG MOVE "Handbook on the external costs of transport" (EC, 2019) not only presents costs per type of transport mode or vehicle and per activity unit for air pollution, greenhouse gases and noise, it also provides costs aggregated over all modes expressed in €/tonne of air pollutant and in €/tonne of CO_2-equivalent. CE Delft being one of the authors, approaches are similar to those in CE Delft (2018). Costs for air pollutants include impacts on health, crop losses (from ozone and other acidic air pollutants such as SO_2, NOx), material and building damage (soiling from particulate matter and corrosion from NOx or SO_2) and biodiversity loss from acidification of soil, precipitation and water (e.g. by NOx, SO_2) and eutrophication of ecosystems (e.g. by NOx, NH_3). Air pollutant damage costs are differentiated as follows: NOx (cities, rural), NMVOCs (all areas), SO_2 (all areas), PM exhaust (metropolitan, urban, rural) and PM non-exhaust (average). These data differ by country. Estimates are based on the impact-pathway approach (modelled with EcoSense). Damage per tonne cost factors for CO_2-equivalents are aggregated over all modes and equal for all countries. Climate change costs are defined as the costs associated with all of the effects of global warming, such as sea level rise, biodiversity loss, water management issues, more and more frequent weather extremes and crop failures[29].

5.2.1 Additional pollutants suggested

Pollutant screening in chapter 3 suggested including CO into future EEA externality assessments. Hospital admissions due to a chronic exposure to CO are considered a relevant health effect but generally agreed response functions are yet lacking. Studies currently quantifying and monetising health impacts from CO

[29] However, EC (2019) uses the abatement cost approach, and not the damage cost approach, for the monetisation of CO_2 (cf. chapter **6**).

are limited to acute deaths from short term (in-door) exposure (poisoning) (Logue et al., 2012; Kopp et al., 2014).

5.2.2 New scientific knowledge relative to impacts on materials

Damage to materials takes several forms:

- Soiling of buildings from deposition of particulate matter
- Corrosion of acid-sensitive stone materials (including cement and mortar), steel, zinc and copper from acidic deposition
- Damage to polymeric materials (e.g. paints, rubbers) from exposure to ozone.

The past quantification for the EEA damage costs focused on effects associated with SO_2 exposure, which has been the primary cause of acidification damage (ExternE, 1998b). However, it is noted that the large reduction in emissions of sulphur in urban areas (reflecting reduced coal burning, cleaner transport fuels, etc.) has seen an end to the rapid and extensive damage to stonework that was a feature of the early to mid-20th century. The full analysis of materials damage costs involved use of the damage functions developed under UNECE International Collaborative Programme (ICP) on Materials and reported in the 2005 update to the ExternE methodology (ExternE, 2005), which also describes the cost data adopted, using guidance from publications for quantity surveyors. Information on the stock of materials at risk comes from earlier reports in the ExternE series (1995, 1998), again drawing on ICP Materials. Later work by ICP Materials has largely focused on damage to buildings of cultural interest. However, there is no inventory to describe the stock at risk for these buildings and structures and so possible damage has gone unquantified. This year, the ICP Materials[30] of the Convention on Long-Range Transboundary Air Pollution (CLRTAP)[31] will publish a first assessment of the direct costs (regarding maintenance, cleaning...) of air pollution on cultural heritage, analysing about 20 UNESCO monuments. To what extent this analysis will allow extrapolation to all buildings of cultural interest remains to be analysed. Also, the issue of lacking stock at risk data persists. There are clear uncertainties in quantification of materials damage, but the magnitude of damage relative to health impacts in particular, suggests that a broad order of magnitude estimate of materials damage is adequate for current purposes. It is, however, recommended that more recent ICP Materials publications[32] are systematically reviewed to ensure that the damage costs that are used reflect current understanding.

A simple approach was adopted for the EEA damage cost assessment, extrapolating from earlier analysis. Damage estimates were small compared to health impacts. No account was taken of the impacts of soiling from particulate matter in the EEA (2014) report. Methods are available to incorporate this element into the analysis. For future work, impacts of soiling from the deposition of particles, therefore, could be brought in, though again, it will not add significantly (more than a few percent) to the health impacts. Ozone effects could also be brought into the analysis, though again this will add little to the total damage estimates. For some materials (different types of stone, galvanised steel, ironwork) the response functions developed through the 1980s to the 2000s likely remain reliable given that the same materials are in use. For some other materials, however, notably paints, they will not be reliable because formulations have changed significantly.

There has not been further work on the overall inventory of acid-sensitive materials at risk since the 1990s and that can be used in Europe wide assessments.[33] The emergence of new materials and design trends

[30] http://www.corr-institute.se/icp-materials/web/page.aspx
[31] http://www.unece.org/env/lrtap/welcome.html.html
[32] http://www.corr-institute.se/icp-materials/web/page.aspx?refid=11
[33] A recent call for data (http://www.corr-institute.se/icp-materials/web/page.aspx?refid=20) yielded qualitative and quantitative data (materials, level of pollutants, meteo-climatic parameters) for in total twenty-one unique cultural objects located at UNESCO World Cultural Heritage sites. A wide range of materials (natural stone, artificial stone, copper, bronze, glass and others) and environmental conditions is represented in the selected sites.

will make the earlier inventories unreliable over time. However, given the longevity of the building stock this may take many years before it has a substantial effect on materials damage costs for air pollution.

Original research of ozone impacts in the UK found that effects were most significant on rubber goods which are chiefly used in vehicles during the 2000s (Holland et al., 2007), but that paint had become significantly more ozone resistant over time. Assuming that vehicle rubber formulations have not changed substantially in the last 15 years, it will remain acceptable to apply the relationships obtained from the UK more widely across Europe, extrapolated according to the size of vehicle fleets and accounting for inflation.

There has long been concern over effects on cultural heritage – indeed, back in the 1970s and 1980s this was regarded as one of the main reasons for reducing European emissions of SO_2 and other air pollutants. However, analysis still lacks a suitable inventory of the stock at risk and information on repair costs.

Given the low contribution to overall damage costs and the lack of new data, only minor revisions to the methods for material damage assessment seem justifiable. The effort required for inclusion of these effects is small, so it is worth retaining them in the analysis.

5.2.3 New scientific knowledge relative to impacts on crops and forests

The 2014 analysis for the EEA included only a limited assessment of crop damage for these receptors. This analysis was based on the AOT40 indicator. Over the last years, various studies have used the more recent PODy indicator to assess damage from ozone to crops and forests (Mills & Harmens, 2011; Anav et al., 2015; cf. also Castell & Le Thiec, 2016, for an overview of recent studies). While the studies cited assess ozone damage for a limited number of crop or tree species, the analysis of crop impacts in the ECLAIRE study (Holland et al, 2015a, b) accounts for damage from exposure to ozone for all of the major European crop species. Similarly, analysis of effects of ozone on timber production and carbon sequestration were quantified for the major European tree species.

The ECLAIRE analysis included use of response functions based on ozone flux measurements, rather than the concentration based AOT40 metric used for previous analysis. The flux-based approach is the one currently supported by science as it produces results that coincide better with observations of ozone damage on vegetation than the results of the metric AOT40 (Hayes et al., 2007). Latest response functions supported by the Convention on Long-Range Transboundary Air pollution are published in LRTAP Convention (2017).

Ineris has developed an offline tool for the calculation of phytotoxic ozone fluxes (PODy) that has been used in a previous ETC report to assess long term trends in the ozone impact on crop yields (Colette et al., 2018). The tool is now being tested to apply to regular ETC Mapping activities, as for instance the EEA Air Quality Report. However, in order to be used in the present context, it would require hourly ozone information, which is not available in the SRMs used for dispersion modelling. Furthermore, SRMs for ozone fluxes are currently not produced by EMEP (cf. chapter 4). At this state, it remains therefore preferable to continue using the AOT40 approach.

With respect to limitations, changes in production of livestock and milk via changes in the quality and quantity of pasture grass is not accounted for in the ECLAIRE analysis. Valuation was performed in a simple way that assumed that there would not be a change in crop or timber price in response to impacts. Holland et al. (2015a) also highlighted potential for significant variation in future estimates of crop damage linked to ozone exposure under alternative climate scenarios.

Overall, at the European scale, damage to forests and crops was equivalent to only about 2.5% of the health impacts. This estimate should be compared to that of other projects - the percentage might be higher for individual countries. In a recent study in France (Schucht et al., forthcoming), the crop damage (wheat, tomatoes, potatoes, grass land) from ozone is estimated to amount to 8% of health costs in 2030. In this analysis, ozone damage on crops is more than three times higher than ozone damage on health. Whatever the exact value, the fact that health costs remain dominant compared to effects on crops might also be explained by the difference in monetisation. While the monetisation of effects on crops is based on product prices, monetisation of health effects also incorporates non-market assessments (willingness

to pay to avoid death, suffering …), which tend to be higher than values estimated based on market valuation.

If PODy source-receptor matrices became available, the assessment of crops and forest impacts should use this indicator. Results could then also be scaled from the ECLAIRE analysis on the basis of estimates of POD (phytotoxic ozone dose). However, in the absence of PODy SRMs, impacts on crops and forests will have to be calculated based on the AOT40 indicator.

5.2.4 New scientific knowledge relative to impacts on ecosystems

ECLAIRE also considered damage to natural ecosystems, chiefly through exceedance of the critical loads for nitrogen for natural terrestrial ecosystems (exceedance for this pollutant occurs across much of Europe). Three methods were used for the analysis:

- Use of repair costs (Ott, 2006)
- Assumption of 'regulatory revealed preference', relating the costs of earlier air pollution action under the UNECE Convention to anticipated ecological improvement
- Use of a UK willingness to pay study (Christie et al., 2012), assessing response to the UK's biodiversity action plan.

Of the three methods, the third approach is considered here to be the most robust, even though it does not account for differences in preference between countries (given that studies similar to Christie et al. have not been performed elsewhere). The repair cost approach assumes that repair is possible, and is based around the use of costings that are clearly very simplified considering the diversity of ecosystems at risk and variation in the extent of damage. The regulatory revealed preference approach makes assumptions of the rationale actually used by the policy makers in development of early protocols to the UNECE Convention. Some comfort can perhaps be gained from the fact that all three methods generate estimates of a similar order of magnitude.

Additional work on ecosystem impacts has been carried out in the UK (Jones et al., 2019). However, the Jones analysis is very specific to the UK situation and could not easily be extrapolated to the EU scale.

Overall, quantified estimates of ecosystem damages are low compared to health impacts, in the order of 2% for assessments consistent with the current methods used for EU policy assessment (Holland et al., 2015b and see

Table 9). For individual pollutants the results may be higher (Ricardo, 2019), depending on what health and other endpoints are included. UK analysis is broadly consistent with this, though there are significant differences between pollutants. Combining data from Defra (2019) and Ricardo (2019) indicates less than 1% of damage attributable to NOx and SO_2 emissions is linked to ecosystems. For NH_3 benefit equivalent to 8% of the damage cost is derived, though this is made up of a disbenefit to ecosystems countered by increased agricultural and forest production. For VOCs, the Defra estimates may include a rather high fraction of damage attributable to ecosystems, but this is entirely down to effects on agriculture and forestry, combined with a low estimate of health damage compared to that obtained using the HRAPIE functions. Significant effort to refine the 2% estimate seems unlikely to yield numbers that are demonstrably more robust given a lack of further valuation data or analysis that links pollutant exposure to valued criteria.

Table 9: Estimated damage for 2010, €billion/year, EU (source: Holland et al., 2015b).

Crops	7.9
Forest,climate	1.1-14
Forestproduction	2.1-3.3
Biodiversity(WTP)	3.8-11
Biodiversity(repaircost)	12
Biodiversity(Reg.rev.pref)	n/a
Health	320
Health(range)	240-1200

The rather small damage estimates for natural ecosystem results appear a weak reflection of willingness to pay given estimated levels of exceedance of the critical load for nitrogen, and the effects that this could have on European ecosystems. However, further detailed research would be needed to substantiate this view. On the other hand, it is not so surprising that willingness to pay for ecosystem protection is significantly smaller than for health protection, with health impacts being more easily understandable to the public and general acceptance that one is responsible for one's own health. It is possible that participants in WTP surveys consider that government should carry a large share of the responsibility for ensuring that ecosystems are protected, particularly when it comes to ecosystems that are in remote locations that few people will access.

Furthermore, experts are also still prudent when establishing linkages between critical loads and biodiversity.

5.3 Greenhouse gases

5.3.1 Assess emissions or impacts?

There are two approaches to deriving costs factors for greenhouse gas emissions:

- Quantify impacts associated with climate change following an IPA approach for future scenarios. Such estimates exist, from various authors but are prone to significant uncertainty, reflecting differences in possible scenarios of the future (economic growth, population growth, climate sensitivity, etc.) and the range of impacts considered. The largest potential impacts also tend to be associated with the most uncertain aspects, for example relating to the possibility of conflict and extreme climatic effects.
- Direct monetisation of emissions using the marginal costs of greenhouse gas abatement. The logic for this approach is in part based on the assumption that climate damage will be prevented by legislation to curb emissions. Under this assumption, an increase in emission from one source would need to be countered by a reduction in another to the extent that overall emissions would remain at the legislated level and hence there would be no net impact from these changes on climate: the only change in cost would then relate to the costs of controlling emissions. Under current trajectories for emissions, the assumption of validity of the marginal costs approach is clearly questionable. Values based on marginal abatement costs are also sensitive to assumptions in the scenarios from which they are derived.

Both approaches, therefore, have their uncertainties. Available data and country choices are further discussed in chapter **6.7**.

It should be noted that much work on the benefits of climate mitigation has focused on the benefits of co-reduction of local/regional air pollutants ($PM_{2.5}$, SO_2, NOx) rather than on those of reduced emissions of the greenhouse gases (e.g. Vandyck et al., 2018; Rafaj et al., 2012; Xie et al., 2018; European Commission, 2011; Schucht et al., 2015). The attraction of valuation using this approach to policy makers working on

climate is that the air pollution co-benefits of climate action are both significant and well researched. They also provide benefit to the region where emissions are reduced and in the short to medium term in contrast to climate benefits that may be experienced in locations remote from the control activities and in the distant future. However, adoption of this approach here would clearly risk the double counting of benefits of other emission changes and so is not recommended.

5.3.2 New scientific knowledge

We are not aware of any fundamentally new approaches to estimating carbon prices. Approaches taken in several recent reports and national guidance books are presented in chapter **6.7**.

5.3.3 Greenhouse gases to cover

It is easy to include any greenhouse gas for which an estimate of CO_2 equivalence can be made. This would include CO_2, CH_4, N_2O, sulfur hexafluoride, 'greenhouse gases' (as an aggregated entry to the E-PRTR) and a number of other pollutants, such as the CFCs and HFCs. A question arises regarding the inclusion or exclusion of CO_2 from biomass combustion: the assumption often applied is that biomass would decay, releasing CO_2 and so its sustainable use is contained within the natural carbon cycle. There are at least two further issues to consider, however. The first is that promotion of renewable energy technologies has led to a high level of industrialisation of biomass burning. Some old coal-fired power stations (e.g. in Belgium, Poland and the UK) have been converted to burn biomass in large quantities, in the order of several million tonnes each year. In the case of the Drax plant in the UK, wood is imported from the Southern US States. Further debate is needed on the sustainability of these activities. A second issue concerns the timescales for release. Natural decay of wood will occur over perhaps many years, whilst combustion releases the CO_2 instantaneously (Johnson, 2009; Cherubini et al., 2011).

5.4 Toxic metals and organics

5.4.1 Limited reporting

For the metals and individual organics, E-PRTR reporting seems likely to be incomplete, with data provided for some facilities, but not all. One reason for this is that emissions for many facilities do not exceed the threshold for reporting. Another may simply be that facilities have not measured emissions. This will make the comparison of facilities with respect to these emissions difficult. However, quantification for sites where data are provided will, at least, provide an indication of the potential magnitude of damage. Results will need to be reviewed and consideration given to a more comprehensive collection of emissions data in future.

Review of results will need to consider the consequences of possible inconsistencies in reporting when reporting and when ranking facilities.

5.4.2 Availability of response functions

A review of the substances that are included in the E-PRTR but for which quantification was not performed in EEA (2014) has been made, with respect to the availability of functions for quantification of health impacts, drawing largely on information from the USEPA's IRIS (Integrated Risk Information System) database. Results are shown in Table 10. The table simply reports the existence of response function data: detailed review of the USEPA information has not been undertaken here and the validity of using specific functions in the present context of industrial emissions of air pollutants is not guaranteed.

Table 10: Screening of substances included in the E-PRTR for which quantification was not made in EEA (2014) relative to availability of data on response functions and valuation.

Substance	Effect	Functions available?
1,1,1-trichloroethane	Non cancer	Yes (USEPA IRIS database)
1,1,2,2-tetrachloroethane	Cancer/non cancer	Yes (USEPA IRIS database)
1,2-dichloroethane (DCE)	Cancers	Yes (USEPA IRIS database)
Aldrin	Cancer/non cancer	Yes (USEPA IRIS database)
Anthracene	?	None identified
Asbestos	Cancer	Yes (USEPA IRIS database)
Carbon monoxide (CO)	Non cancer	Yes, but threshold issues
Chlorine and inorganic compounds	Non cancer	None identified, threshold issues
Chlorofluorocarbons	Ozone layer, climate	For climate
Copper	?	None identified, threshold issues
Di-(2-ethyl hexyl) phthalate	Possible carcinogen	None identified
Dichloromethane	Cancer	Yes (USEPA IRIS database)
Ethylene oxide	Cancer	Yes (USEPA IRIS database)
Fluoranthene	Non cancer	Yes (USEPA IRIS database)
Fluorine and inorganic compounds	Non cancer	None identified, threshold issues
Halons	Cancer/non cancer	Difficult to treat as a group
Hexachlorobenzene	Cancer/non cancer	Yes (USEPA IRIS database)
Hydrochlorofluorocarbons	Ozone layer, climate	For climate
Hydro-fluorocarbons	Climate	Yes
Hydrogen cyanide	Noncancer	Yes (USEPA IRIS database)
Methane	Climate, ozone	Yes
Naphthalene	Non cancer	Yes (USEPA IRIS database)
Nitrous oxide	Climate	Yes
Perfluorocarbons	?	
Phenols	Non-cancer	Difficult to treat as a group
Polychlorinated biphenyls	Cancer	Yes (USEPA IRIS database)
Polycyclic aromatic hydrocarbons	Cancer	Yes
Sulphur hexafluoride	Climate	Yes
Tetrachloroethylene (PER)	Cancer/non cancer	Yes (USEPA IRIS database)
Tetrachloromethane (TCM)	Cancer/non cancer	Yes (USEPA IRIS database)
Trichlorobenzenes (TCBs) (all isomers)	Non cancer	Yes (USEPA IRIS database)
Trichloroethylene	Cancer/non cancer	Yes (USEPA IRIS database)
Trichloromethane	?	
Vinyl chloride	Cancer/non cancer	Yes (USEPA IRIS database)
Zinc and compounds (as Zn)	Non cancer	No

In some cases, toxicity has been reported, but subject to thresholds that may be unlikely to be exceeded via ambient exposures.

The table indicates that there is scope for quantification of effects for most of these substances. However, the IRIS database provides unit risk factors or slope factors that can be used in a lifetime "risk" analysis. This is different from an "impact" analysis. Furthermore, the IRIS response functions are often based on experimentation animal data, and when they consider health outcomes other than cancer contain a threshold. This excludes the possibility to quantitatively assess health damages. Studies of health effects of vinyl chloride are limited to animal tests and epidemiological studies for workers. We have not found any dose-response functions based on epidemiological data and applicable to the general population.

Concerning specifically the pollutants whose addition was suggested as a result of the pollutant screening (chapter 3), no response functions have been identified for fluorine and chlorine. Response functions for

vinyl chloride and 1,2-dichloroethane are limited to those discussed in the previous paragraph. We have no information on dichloromethane.

Copper and zinc are amongst the list of pollutants considered as "emerging" by the WHO (2013b). In France, ANSES (2018) considers copper as a priority substance for a potential future ambient air quality surveillance. Zinc is not considered as priority pollutant. We are not aware of broadly agreed dose-response functions for copper and zinc that could directly be integrated into the analysis. The shadow price for emissions of zinc (health impact) estimated by CE Delft (2010 a & b) for the Netherlands and CE Delft (2018) for the EU28, is also very small compared to that of arsenic and lead (cf. Table 11). For all these reasons we consider that the inclusions of these two elements in the upcoming EEA assessment is premature and not a priority.

Table 11: Damage costs for atmospheric emissions (€/kg, 2016 prices)

	The Netherlands	EU28
Copper	0,4	3,9
Zinc	11,3	6,7
Arsenic	899	864
Lead	452	5383

Sources: CE Delft (2010), CE Delft (2018)

5.4.3 Comparison between modelling frameworks for toxic metals

Data are available in terms of damage per tonne of emission for two models for a selection of metals (cadmium, chromium and nickel): for the simplified modelling framework used in EEA (2014) and the more complex system adopted in the ESPREME model. For cadmium and chromium, EEA results were a factor 3.3 greater than ESPREME, whilst results for nickel were a factor 2.0 greater (Table 12). However, adjusting response and valuation functions reduced the difference between the results of the two models to less than 10% in all cases. This suggests that the most sensitive factor in deriving damage costs for these pollutants lies in the impact assessment and valuation phases rather than exposure assessment, and hence that the more complex models are of at best limited advantage in externality assessment relative to the simple tool used previously.

Table 12: Comparison of EU averaged results from ESPREME and EEA (2014), showing the ratio of the results and changes to the ratio following adjustment of response functions and valuation factors

	ESPREME	EEA(2014)	EEA/ESPREME	Adjust response	Adjust value
Cd	7,123	23,093	3.24	3.24	1.03
Cr	9,230	30,936	3.35	3.35	1.07
Ni	1,549	3,100	2.30	3.17	1.01

5.4.4 New scientific knowledge relative to health impacts

Past analysis of the impacts of metals and (persistent) organics has focused on one (or at best a small number) of the potential impacts of those pollutants. Hence, for lead and mercury, analysis focused on loss of IQ from exposure when young, whilst for other pollutants consideration has typically gone to quantification of cancer impacts. The focus on these endpoints is understandable from the perspective that they have been well researched and are clearly serious effects.

The problem of reliance on only one or two effects per pollutant was demonstrated in two papers by Nedellec and Rabl (2016a, b; see also Nedellec et al., 2019). Concerned that estimates of the benefits of

reducing these pollutants could be underestimated, they reviewed evidence for a much wider range of health impacts than is typically quantified (previous estimates being focused only on cancer deaths and IQ loss):

- Arsenic: Non-cancer mortality, cancer deaths, non-fatal cancers, chronic bronchitis, IQ loss, infant deaths, diabetes
- Cadmium: Mortality, non-fatal cancers, fractures
- Lead: Mortality, IQ Loss, anaemia
- Mercury: Mortality, IQ loss

To illustrate, IQ loss linked to lead exposure was estimated to have a cost of €4,435/kg, whilst mortality risks were about 6 times higher at over €24,858/kg ($€_{2013}$)[34] acknowledging significant uncertainty with a geometric standard deviation of 4. For mercury the situation was more extreme with a factor 10 difference between impacts on IQ (€2,100/kg) and on mortality (€21,000/kg)[35]. Similar patterns were found for the other two pollutants considered, arsenic and cadmium although analysis of those substances dealt with a different set of impacts.

Nedellec and Rabl (2016a) considered consequences of their results for waste incinerators in France. Using actual emission data, results for toxic metals were about half of those for PM_{10}, SO_2 and NOx. The effects of toxic metals were estimated to exceed those of PM_{10}, SO_2 and NOx for a hypothetical plant operating at the EU emission limit values. Results demonstrate that the assumptions on which impacts should be included can have a significant influence on total externalities.

Per unit mass emitted, the damage costs associated with most toxic metals and some organics have been far higher than for what we might call the major regional air pollutants (NH_3, NOx, $PM_{2.5}$ and SO_2). However, the mass emitted is relatively small, such that most damage is linked to the major air pollutants. Analysis provided in the Nedellec and Rabl (2016 a & b) papers demonstrates that there is potential for the impacts of the trace metals to reach a level comparable with other pollutants for some sectors. This should be considered in the identification of the pollutants for which damage costs are required, irrespective of the number of plants for which data are currently being provided.

When considering including additional health effects from toxic metals, attention should be paid to the risk of double counting. Some risk of double counting exists when calculating the mortality externalities of metals on top of chronic mortality from $PM_{2.5}$. However, some consideration needs to be given to the way that the damage costs will be used in future, remembering that they have been used extensively. In some cases, the damage costs will be used to generate aggregate damage for sectors or facilities, and in this case there is certainly potential for double counting. However, if the question was different, regarding how much damage is linked specifically to metal emissions there is no risk of double counting. Two sets of damage costs (with and without mortality effects) should be generated to provide figures that are relevant to both types of question.

There are further recent studies compiling evidence of health effects of toxic metals and several studies focus on cardio vascular disease and mercury. Hu et al. (2018), for example, review the association between mercury exposure and the prevalence of hypertension, which they conclude to be non-linear. They also conclude that more systematic review is needed to better qualify this relationship. Solenkova et al. (2014) review the evidence of mercury, lead, cadmium, and arsenic, having epidemiologic and mechanistic links to atherosclerosis and cardio vascular disease (CVD).

There is clearly a need to review the logic for defining which impacts should be attributed to the trace metals and organic pollutants. If results are considered to be too uncertain it may be appropriate to define sensitivity scenarios to consider under what assumptions the damage associated with emissions of these pollutants becomes significant relative to other pollutant externalities.

[34] In 2016 prices € 4475 and € 25083, respectively.
[35] In 2016 prices € 2123 and € 21022, respectively.

5.5 Conclusions taking account of feedback from the expert consultation

We support the earlier approach of providing country specific unit damage costs for emissions of the major air pollutants that cover the impact of emission per tonne of pollutant from a particular country, wherever the impacts occur (i.e. specifically including transboundary impacts). Note that this approach is to some extent simplified as we will not cover effects outside the European domain modelled. We suggest to additionally present the damage occurring only in the country.

Gaps identified in the previous assessment (EEA, 2014) that could be filled, based on more recent scientific information relate to:

- Expansion of the health response functions for PM and O_3 exposure if it is considered that additional impacts are sufficiently robust to be included in the analysis;
- Inclusion of NO_2 health impacts due to improvements of approaches to NO_2 exposure modelling;
- Inclusion of soiling of materials from particle deposition and ozone impacts on materials;
- Accounting for additional health impacts from metals, such as cancer or mortality from mercury and lead.

Concerning health impacts, the damage cost analysis by Ricardo (2019) for DEFRA assessed productivity impacts from air pollution ($PM_{2.5}$, ozone) in a more comprehensive way than what was included in the previous assessment. There is also ongoing work in this area by OECD. An update of the previous approach is an objective for the damage cost assessment in 2020. For the other effects included in the Ricardo or Danish analyses, but not in previous EEA assessments (such as cardio vascular disease or stroke), it is possible to apply them to derive an upper bound estimate of damage costs in the 2020 assessment. Still, we consider the evidence base as too uncertain for inclusion in the best estimates at the present time.

There are a few publications suggesting updated response functions for mortality effects from $PM_{2.5}$. Unlike the HRAPIE function these are non-linear. In the absence of a review of response functions by the WHO, we suggest continuing using the HRAPIE (WHO, 2013a) response functions for all-cause mortality. The possibility of a side evaluation (sensitivity calculation) with a more recent response function (such as used in Burnett et al. (2018), Lelieveld et al. (2019) or Pascal et al. (2016)) will be considered in the analysis in 2020.

Health impacts from NO_2 were not included in the previous assessment. However, on a national level, effects from this pollutant are considered (France, UK). We suggest extending future EEA assessments to include health impacts from NO_2[36]. However, different exposure-response functions are currently used for assessing NO_2 mortality. Previous assessments for EEA have been based on the HRAPIE approach (e.g. EEA, 2018). The Ricardo (2019) study relies on more recent guidance from COMEAP. For the EEA assessment, we could follow HRAPIE in the absence of new WHO guidance, but, as stated in chapter 4, the use of a response-function with a threshold together with available mid-resolution source-receptor matrices will involve high uncertainty, tending to underestimate mortality effects from NO_2. The consulted experts agreed on this uncertainty being too high. For the 2020 analysis we suggest using source receptor matrices derived from the CHIMERE-SHERPA exposure model (7km spatial resolution) for no-threshold response functions. Therefore, if the WHO proposes a new response function without a cut-off point for chronic mortality, we will use this function. Otherwise, we will not include NO_2 chronic mortality into this year's damage cost assessment. However, a screening analysis to assess sensitivity of results to different available response functions for NO_2 chronic mortality on overall health damage will be carried out based on high resolution chemistry transport modelling in France. In this screening we will compare the HRAPIE function with the different COMEAP (2018) views.

We suggest keeping the impacts of air pollution on materials in future analyses. In the past, only corrosion effects on materials were accounted for. The impacts of soiling from particulate matter and impacts of ozone could be brought in the analysis, even though this will not add significantly to the health impacts.

[36] Inclusion of NO_2 health effects as suggested by HRAPIE would not be limited by economic valuation. Recommended health impacts are all regularly assessed for $PM_{2.5}$ and/or ozone, so unit values exist.

Furthermore, information on updated response functions for materials that may have changed since the currently used functions were developed (e.g. paints) should be collected in the 2020 assessment.

For trace metals and organic pollutants there is still an issue of which health impacts should be included in the assessment, given uncertainty in impacts. Recent analyses (Nedellec and Rabl, 2016 a, b) demonstrate that there is potential for the impacts of the trace metals to reach a level comparable with other pollutants for some sectors. This should be considered in the identification of the pollutants for which damage costs are required, irrespective of the number of plants for which data are currently being provided. However, the fact that these authors assess mortality for the birth cohort rather than for the population alive as is the case for other health impacts assessed in earlier EEA reports would require refinement of the Nedellec and Rabl (2016 a, b) results before they can be included in the updated externalities assessment. Furthermore, attention will need to be paid to the risk of double counting, for example when chronic mortality not only of PM but also of mercury and lead is taken into account. Further assessment is therefore required before additional impacts of metals can be included in the analysis, as noted.

In the last EEA (2014) report, impacts on crops and forests were assessed based on the earlier indicator AOT40. The scientifically recommended indicator is the stomatal ozone flux, we would therefore, in principle, recommend using it in future analyses. Advances in the estimate of crop and forest damage using the ozone flux concept have been made since the EEA (2014) report. New scientific information relates also to the inclusion of more crop varieties in assessments of impacts from ozone and climate change (ECLAIRE project) than based on the response functions published in LRTAP (2017). However, as long as no PODy SRMs are available, the calculation of impacts of ozone on crops and forests will need to continue using the AOT40 indicator.

High levels of uncertainty preclude adoption of some impacts for the analysis:

- For cultural heritage, comprehensive inventories are still lacking.
- Changes in production of livestock and milk via changes in the quality and quantity of pasture grass as a result of ozone impacts is not included in the earlier assessment. No new information is available on these impacts. Also, we do not expect them to significantly change the overall results.

Uncertainty in quantifying ecosystems and biodiversity impacts is still high. Nevertheless, biodiversity effects from deposition of NH_3 and NO_2 were calculated, for example in the ECLAIRE study, for exceedances of critical loads in Natura 2000 areas[37]. The same approach could be applied here, although monetised impacts are likely to remain low. Given the political importance of biodiversity, and the extent of critical loads exceedances for nitrogen, we suggest including such effects in the 2020 EEA assessment.

So far, no direct effects of specific VOCs (often carcinogenic, mutagenic and toxic for reproduction - CMR) have been covered by the assessment. If this was to be amended there would be a need to assess how population exposure could be determined. Inclusion in the 2020 assessment appears premature. However, this issue should be followed-up and the approach should be reviewed in the coming years.

With respect to pollutants whose integration was a suggestion from the screening analysis in Chapter 3, we are not aware of broadly agreed dose-response functions for copper and zinc that could directly be integrated into the analysis. Similarly, no response functions have been identified for fluorine and chlorine. We therefore do not recommend the inclusion of these elements in the upcoming EEA assessment. IRIS response functions are available for vinyl chloride and 1,2-dichloroethane. Studies of health effects of vinyl chloride are limited to animal tests and epidemiological studies for workers. We have not found any dose-response functions based on epidemiological data and applicable to the general population. Finally, we have no information on dichloromethane.

Concerning CO, while hospital admissions due to a chronic exposure to CO are considered a relevant health effect, generally agreed response functions are yet lacking. Studies currently quantifying and monetising

[37] Impacts of acidification were not specifically taken into consideration because exceedances of critical loads for acidification are currently much less important than for eutrophication. The rationale was that including impacts from acidification would not have had an important impact on overall results.

health impacts from CO are limited to deaths from short term (in-door) exposure (cf. chapter 5.2.1). We therefore suggest to not yet include CO into the analysis.

Out of the 8 pollutants and 2 greenhouse gases whose integration was suggested from the screening analysis, we suggest retaining only the two greenhouse gases (methane and nitrous oxide, cf. chapter 6). Note that the contribution of methane to ozone, however, cannot be accounted for as there are currently no SRMs for methane.

The consulted experts agreed with us that to treat biomass use as carbon neutral is simplistic. The main reason for this is the lag between carbon release and carbon sequestration. The long-term sustainability of biomass use can also be questioned. However, there is an unresolved scientific debate on what assumptions to use with respect to the CO2 emissions of biomass combustion. The extremes are (naturally) assumptions that either 0% or 100% of the carbon released from biomass combustion should be considered. For next year's assessment we suggest using a – somewhat arbitrary - middle-option between these extremes. Plausible options are that 50% of the released carbon should be included, or another figure based on emissions discounted over the period taken for full take up by the biosphere, which would be more complex. The final decision must, however, be transparent.

6 Valuation of quantified impacts

6.1 Situation in the previous assessment

The 2014 analysis adopted the same assumptions as those routinely used for quantification of air pollution impacts in policy studies for the European Commission (see, e.g. Holland, 2014). The values for other elements, such as climate change mitigation costs, were also consistent with those used by the Commission services.

Monetary values for different health or non-health end points affected by air pollution seek to capture associated costs in a way as comprehensive as possible. For health end points for example they seek to capture costs to employers, costs of healthcare and willingness to pay to avoid pain, suffering and premature death. The general approach for valuation of impacts therefore combines individual willingness to pay for health (etc.) protection with additional elements, for example the costs of healthcare and lost productivity. For climate change, the preference in previous reports has been to base costs on the marginal cost of GHG mitigation.

6.2 New scientific knowledge

In late 2018 and early 2019 the German Environment Agency (Umweltbundesamt, UBA) published an update of its Convention on methods for determining environmental costs, including cost assessments for emissions of air pollutants and greenhouse gases (UBA, 2018 & 2019). The cost factors averaged at country level are given in Table 13. Quantification and monetization of impacts is based on HRAPIE (WHO, 2013a) and Holland (2014) assumptions, otherwise costs were estimated based on updated NEEDS (2008) data.

Where cost factors for impacts were initially expressed for other years, the update to current and future years is based on the German consumer price index. For data representing willingness to pay (WTP) to avoid impacts it was assumed that they should increase with income. Cost factors were corrected by the growth in per capita GDP from source to target year, also using an income elasticity of 0,85. The report does not indicate whether the unit cost factors account for the damage occurring only in Germany or across the EU28.

Table 13: UBA recommendations for average environmental costs of air pollutant in Germany, in € per tonne of emission (€ price base 2016)

Tabelle 2: Durchschnittliche Umweltkosten der Luftverschmutzung durch Emissionen aus unbekannter Quelle (in $€_{2016}$ / t Emission)

	Kostensätze für Emissionen in Deutschland				
$€_{2016}$/t Emission	Gesundheits-schäden	Biodiversitäts-verluste	Ernte-schäden	Material-schäden	Gesamt
Deutschland gesamt					
$PM_{2.5}$	58.400	0	0	0	**58.400**
PM_{coarse}	960	0	0	0	**960**
PM_{10}	41.200	0	0	0	**41.200**
NO_X	14.400	2.600	800	130	**17.930**
SO_2	13.600	1.000	-160	600	**15.040**
NMVOC	1.100	0	950	0	**2.050**
NH_3	21.700	10.400	-100	0	**32.000**

Annahme: PM_{10} besteht zu 70% aus $PM_{2.5}$ und zu 30% aus PM_{coarse}. Für NO_x und SO_2 bilden die Kosten die Schäden durch sekundäre Feinstaubbildung ab. Quelle: Van der Kamp et al. (2017).

Source: UBA (2019) - "Gesundheitsschäden" = health damage; "Biodiversitätsverluste" = loss in biodiversity, "Ernteschäden" = crop damage; "Gesamt" = Total (damage cost)

UBA (2019) recommends using such averaged costs per tonne of pollutant emitted for emissions from "unknown sources", when no specific information on emission sources is available. For location specific assessments, they recommend cost factors differentiated by sector and population density (urban vs. rural areas). UBA considers this most important for particulate matter emissions, for which impacts differ importantly as a function of the emission source (source height).[38]

Table 14: UBA recommendations for power stations, industrial combustion and small combustion plants in Germany, in € per tonne of emission (€ price base 2016)

Tabelle 3: Kostensätze für die Emission von Luftschadstoffen aus Kleinfeuerungsanlagen und Verbrennungsprozessen in der Industrie (in €$_{2016}$ / t Emission)

	Gesundheitsschäden											Materi al- schä- den	Ernte- ausfälle	Biodi- versität
	Kraft- werke	Verbrennungsprozesse in der Industrie					Kleinfeuerungsanlagen							
		Unbe- kannt	Großstadt		Stadt		Unbe- kannt	Großstadt		Stadt				
Höhe (in m)	>100		0-20	20-100	0-20	20-100		0-20	20-100	0-20	20-100			
$PM_{2.5}$	31.500	64.900	116.100	65.500	80.400	65.500	61.700	110.400	62.300	76.500	62.300	0	0	0
PM_{coarse}	400	1.100	2.000	1.100	1.400	1.100	1.000	1.800	1.000	1.300	1.000	0	0	0
PM_{10}	22.100	45.700	81.900	46.200	56.700	46.200	43.500	77.800	43.900	53.900	43.900	0	0	0
NO_x	11.100	15.200	15.200	15.200	15.200	15.200	15.800	15.800	15.800	15.800	15.800	100	800	2.600
SO_2	12.700	14.500	14.500	14.500	14.500	14.500	14.700	14.700	14.700	14.700	14.700	600	-200	1.000
NMVOC	1.200	1.200	1.200	1.200	1.200	1.200	1.200	1.200	1.200	1.200	1.200	0	1000	0
NH_3	23.700	23.700	23.700	23.700	23.700	23.700	23.600	23.600	23.600	23.600	23.600	0	-100	10.400

Kategorien „Großstadt" und „Stadt" unterscheiden sich nach der Gemeindegröße (Großstadt >100.000, 2.000<Stadt<100.000) Annahme: PM_{10} besteht zu 70% aus $PM_{2.5}$ und zu 30% aus PM_{coarse}. Diese Annahme sollte angepasst werden, falls quellenspezifische Informationen zur Zusammensetzung vorliegen. Für NO_x und SO_2 bilden die Kosten die Schäden durch sekundäre Feinstaubbildung ab. Quelle: Van der Kamp et al. (2017) und eigene Berechnungen.

Source: UBA (2019) – "Gesundheitsschäden" = health damage; "Kraftwerk" = power station/energy production; "Verbrennungsprozesse in der Industrie" = industrial energy production; "Kleinfeuerungsanlagen" = small combustion plants; "unbekannt" = unknown location (= average); "Grossstadt" = big city (> 100,000 inhabitants); "Stadt" = city (>2,000 but < 100,000 inhabitants); "Höhe (in m)" = height in meters (of emission source)

It is noted here that apart from PM, there is little variation across the different source types. This is partly due to the importance of the secondary pollutants in causing the damage associated with these emissions. For PM, the difference across sites seems small, especially when compared with the variability observed in the Defra (2019) estimates from the UK. The differences are clearly associated with the dispersion modelling assumptions. Despite this observation, it is noted that results for Germany and the UK are of a broadly similar magnitude (to the extent that comparison is possible), except for NMVOC where the UK results are very low.

Industry specific damage costs are given in Table 14 that can be set in relation to the average cost presented in Table 13. The result is presented in Table 15 (sector specific damage per tonne estimates were divided by the average damage per tonne estimate to calculate "potencies" (cf. chapter 4)). The exact link of the indicated sectors with SNAP nomenclature categories is not specified. We assume the following: power plants belong to SNAP1, small combustion plants to SNAP2 and industrial energy production to SNAP3.

[38] Values for transport emissions are given in the report (UBA, 2019).

Table 15: Industry specific shares in average country damage costs in UBA (2019b)

UBA (2019b)	€/tonne of pollutant				Average unit damage cost ratio (potency)		
Damage costs for Germany	Average Germany	Power plant	Industrial combustion processes (not location specific)	Small combustion plants (not location specific)	Power plant	Industrial combustion processes (not location specific)	Small combustion plants (not location specific)
PM2.5	58 400	31 500	64 900	61 700	0,54	1,11	1,06
PM coarse	960	400	1 100	1 000	0,42	1,15	1,04
PM10	41 200	22 100	45 700	43 500	0,54	1,11	1,06
NOx	14 400	11 100	15 200	15 800	0,77	1,06	1,10
SO2	13 600	12 700	15 400	14 700	0,93	1,13	1,08
NMVOC	1 100	1 200	1 200	1 200	1,09	1,09	1,09
NH3	21 700	23 700	23 700	23 600	1,09	1,09	1,09

UBA (2019) also presents costs of electricity production per kWh for different energy sources and in this calculation includes emissions from the whole life cycle of the technologies.

Unit values for health and environment impacts used in the Handbook on the External Costs of Transport (EC, 2019) are taken from NEEDS (2008a), Desaigues (2011) and Rabl et al. (2014). Cost factors were updated by taking into account concentration-response functions also from HRAPIE (WHO, 2013a), recent population data, and adjustments for valuation data (details are provided in annexes A and C of the handbook) and for some pollutants cost-factors were differentiated. They are presented in Table 16. Impacts of ozone are accounted for under the respective precursor pollutants. Mortality from air pollution is valued using VOLY (Value of Life Year Lost). The report does not indicate whether impacts occurring across EU-28 (or beyond) are included in the country values or only impacts occurring within the respective country (the latter is likely).

Table 16: Air pollution costs for all transport modes according to the Transport cost handbook (€ price base 2016)

		€/ton NOX	€/ton NOX	€/ton NMVOC	€/ton SO2	€/ton PM (exhaust)			€/ton PM10 non-exhaust
		Transport city	Transport rural	All areas	All areas	Transport metropole	Transport urban	Transport rural	Average
EU Aggregate	EU-28	21 300	12 600	1 200	10 900	381 000	123 000	70 000	22 300
Austria	AT	41 400	24 300	2 300	16 200	466 000	151 000	87 000	30 900
Belgium	BE	26 100	15 100	3 600	17 100	479 000	155 000	114 000	47 200
Bulgaria	BG	10 000	5 900	0	4 200	191 000	61 000	30 000	5 400
Croatia	HR	18 500	11 400	900	8 000	292 000	95 000	54 000	8 200
Cyprus	CY	8 100	4 500	-400	7 800	0	71 000	17 000	20 100
Czech Republic	CZ	24 800	14 800	1 100	11 600	361 000	116 000	72 000	39 600
Denmark	DK	16 200	9 600	1 500	9 600	470 000	151 000	59 000	15 000
Estonia	EE	5 400	3 400	300	5 200	0	102 000	35 000	4 900
Finland	FI	5 300	3 500	400	4 600	366 000	118 000	32 000	11 900
France	FR	27 200	16 200	1 500	13 900	407 000	131 000	87 000	5 900
Germany	DE	36 800	21 600	1 800	16 500	448 000	144 000	93 000	24 700
Greece	EL	5 100	3 100	300	5 900	267 000	86 000	33 000	24 800
Hungary	HU	26 800	15 800	800	9 900	317 000	102 000	59 000	8 500
Ireland	IE	17 600	10 100	1 700	11 800	568 000	183 000	68 000	12 200
Italy	IT	25 400	15 100	1 100	12 700	409 000	132 000	79 000	19 000
Latvia	LV	7 200	4 400	400	4 800	251 000	81 000	28 000	17 200
Lithuania	LT	12 100	7 100	600	6 400	300 000	98 000	38 000	27 000
Luxembourg	LU	66 800	38 400	6 200	29 300	0	278 000	191 000	8 000
Malta	MT	2 300	1 400	400	4 300	0	72 000	18 000	63 900
Netherlands	NL	26 500	15 300	2 800	20 200	458 000	148 000	101 000	5 600
Poland	PL	14 700	8 900	700	8 200	282 000	91 000	52 000	5 200
Portugal	PT	2 800	1 700	500	4 100	292 000	94 000	39 000	47 300
Romania	RO	19 400	11 200	500	7 300	272 000	88 000	42 000	16 100
Slovakia	SK	24 800	14 700	700	10 100	328 000	105 000	59 000	12 300
Slovenia	SI	22 300	13 700	1 200	9 200	0	93 000	52 000	12 000
Spain	ES	8 500	5 100	700	6 800	348 000	112 000	46 000	10 200
Sweden	SE	9 500	6 000	700	5 500	374 000	120 000	38 000	15 200
United Kingdom	UK	13 600	7 900	1 400	10 000	380 000	122 000	65 000	16 200
Norway	NO	29 161	17 250	1 643	14 923	521 605	168 392	95 833	30 530
Switzerland	CH	30 017	17 756	1 691	15 361	536 918	173 336	98 646	31 426

Source: EC (2019)

*Note: **PM10 cost factors** can be used for the non-exhaust emission of particles PM, e.g. from brake and tyre abrasion. **Metropole** only applies to cities larger than 0.5 million inhabitants. Some countries do not have such cities hence these damage values are hence not being reported. This is the case for Slovenia, Malta, Luxembourg, Estonia and Cyprus. **Rural area**: outside cities; **metropolitan area**: cities/agglomeration with more than 0.5 million inhabitants.*

UBA (2018 & 2019) and EC (2019) use bibliographical references and hypotheses rather similar to those in EEA (2014).

CE Delft (2018) proposes environmental damage costs for air pollutants, expressed as average values across EU28. The central estimates are presented in Table 17. They are based on the NEEDS (2008a, b, c) impact pathway and valuation approaches.

Table 17: Central values for EU28 damage costs for air pollutants according to CE Delft (2018)

Damage costs in €/tonne of pollutant (€ price base 2016) for EU28				
$PM_{2.5}$	NOx	SO_2	NH_3	VOC
38 797	14 837	11 529	17 544	1 153

There are some newer data available on valuation since the EEA (2014) assessment was finalised. For morbidity for example, there are new estimates available from UK analysis by Ricardo (2019), indicating that the values used in policy analysis by the European Commission may underestimate the value of some endpoints. New estimates are also available from Denmark (see below).

Further new valuation work compared to EEA (2014), on ecosystems, crops and forests is available via the ECLAIRE study. However, associated values are unlikely to have a significant impact on overall costs given the dominance of health impacts (see Table 9).

6.3 Impacts on health

The most important issues in valuation concern the values of health impacts, especially mortality, given that this makes up a very large share of the total damage costs. For mortality, there has long been a dispute as to whether it is better to value the loss of life years using the value of a life year (VOLY), or deaths using the value of statistical life (VSL). This situation remains unresolved. Some countries and organisations have a clear preference for use of the VSL (e.g. OECD[39], US EPA), whilst others prefer the VOLY (UK government, The Netherlands). Combinations of both are also used, where VSL are used for acute deaths and VOLY for chronic deaths (e.g. EEA, 2013). The European Commission takes results with mortality valued with both the VOLY for lower bound estimates and VSL for upper bound estimates (this range of course does not represent a proper uncertainty analysis). This creates a difficulty in communication for the damage costs with previous estimates of unit damage costs (EEA, 2014) providing a range based around the VOLY-VSL sensitivity, leaving open the question of whether it is preferable to use the upper or lower bound or perhaps some midway point. The range is further complicated by inclusion of both median and mean estimates of the VOLY and VSL, both in the work for the EEA as in the framework of the TSAP. The 2014 assessment used VSL estimates based on the CAFE (Clean Air for Europe) cost-benefit assessment methodology (Hurley et al., 2005; cf. also Holland, 2014) ranging from 1.09 million € (median) to 2.2 (mean) million € in 2005 prices (1.3 – 2.7 million € in 2016 prices) and VOLY ranging from 57.7 k€ to 138,7 k€ in 2005 prices (70 k€ - 168 k€ in 2016 prices).

The VSL values, therefore, remain below what OECD (2012) in its meta-analysis of mortality valuation studies recommends for the EU27 (€2.9 million in 2005 prices, €3.5 million in 2016 prices). The reason is that although this work was funded by the European Commission, its recommendations have not yet been formally accepted for adoption in EC analysis (Ricardo, 2018).

Nevertheless, the DG MOVE Transport cost handbook (EC, 2019) does base the VSL it uses on OECD (2012) and recommends a VSL of € 2.9 million (calculated as an average VSL of € 3.6 million corrected for consumption loss[40], in 2016 prices) for the EU28. For the VOLY the same study uses a value of € 70,000 (€2016), taken from a Dutch study (CE Delft, 2017) and considered as not unreasonable in view of a review

[39] See also Alberini (2017).

[40] In this study the consumption loss of a fatality is calculated by multiplying the annual average consumption expenditure per capita with the number of life years lost. The authors argue that this consumption loss needs to be deducted from the EU28 VSL to reach the EU28 human costs of a fatality. One expert consulted cautioned against applying the consumption loss correction introduced by CE Delft/DG MOVE, this not being common practice.

of different VOLY values suggested[41]. The VSL and VOLY values are differentiated to the individual country level. It should be recalled, however, that mortality from air pollution in the DG MOVE study is only valued on the basis of life years lost (VOLY).

In France, the Quinet report (Quinet, 2013) adopts a value of €3 million (in 2010 prices, €3.2 million in 2016 prices) for valuing a VSL in France, based on OECD (2012). The OECD (2012) value was also used by the World Bank (2016) for the evaluation of health effects from air pollution. For a VOLY, Quinet (2013) suggests € 115 000 (price base 2010, €123 000 in 2016 prices), a value derived from the VSL based on the following relation:

$$VSL = \sum_{t}^{T} VOLY \,.\, (1+\delta)^{-t}$$

With T the number of remaining life years, δ the discount (actualisation) rate, and thus assuming that VOLYs are constant over the remaining life span.

In 2011, Desaigues et al., published the results of a 9 country European study that lead to a VOLY of 43.5 k€ (in 2008 prices) for the EU15.

In Denmark, the Ministry of Finance has published an updated cost for a statistical life in 2017 raising it to 32 million DKK (DCE, 2018), a value around 4.3 million € (price base 2016) which corresponds to the value suggested by OECD (2012) adjusted for the higher Danish GDP/capita.

The cost data used for morbidity in the 2014 report are now rather dated. Some of the figures used, for example, in valuation of hospital admissions seem likely to bias to underestimation, with significantly higher figures being used elsewhere for those effects (Ricardo, 2019, use figures about four times higher for hospital admissions). Data on unit costs for health impacts from European and US sources, including new work (e.g. under the NordicWelfAir study which is due to report in 2020) is currently being collated. Any revision of the figures should be carried out in agreement with European Commission services to ensure that there are not inconsistencies in approach across the Commission and its agencies.

In the context of NordicWelfAir work has been undertaken to update morbidity valuation for health effects supported by HRAPIE (WHO, 2013) based on a consistent methodology. The unit costs of air pollution health impacts are presented in a memo for Denmark's Ministries (Andersen et al., 2019). For use in Europe the values would have to be adjusted from Nordic to EU28 price levels.

The inclusion of additional health endpoints either for the major pollutants or the metals and organics, or for additionally considered pollutants, will require the adoption of new valuation data compared to the data used in the earlier study. A starting point for the new data will be studies such as those of Nedellec and Rabl (2016a, b) and Ricardo (2019), where quantification has already been carried through to monetisation.

Other coordinated actions to assess WTP to avoid health effects are ongoing, such as the OECD SWACHE project (Surveys on Willingness-to-Pay to Avoid Negative Chemicals-Related Health Effects), but the results are not yet available[42]. This project deals for instance with the following health impacts, asthma, IQ loss, low and very low birth weight, kidney failure and fertility loss. Work on monetisation of health effects is also carried out by ECHA (e.g. 2016)[43].

A further factor concerns the severity of illness that can be linked to air pollution exposure. Early monetisation of chronic bronchitis assumed individuals suffering the disease would have significant health problems and associated restrictions on daily activities. However, the definition of chronic bronchitis used in the epidemiological literature laid open the potential for individuals with very mild disease to be included. Given that severe bronchitis is typically associated with heavy smoking, the assumption that

[41] "Taking all of the above into consideration, our literature review has revealed that an EU28 VOLY of € 70,000 (2016 prices) is not unreasonable."

[42] And it is at this stage unclear whether the values that are intended for use in a chemicals policy context will be relevant in the air pollution context.

[43] E.g. for skin irritation, kidney disease and failure, fertility and development toxicity, cancer.

cases of bronchitis quantified as caused by air pollution should be given a very high costing might be questioned. An argument against the concern that too many cases are counted with the current exposure-response function is the fact that (i) there is a huge underreporting of Chronic obstructive pulmonary disease (COPD) and (ii) COPD if detected is usually only done so when getting to moderate or severe stages. Recent information on costs per case and year of bronchitis and COPD will be considered in the calculation of damage costs in 2020 (e.g. Andersen, 2018, Andersen et al., 2019). OECD (2019) published unit costs for bronchitis (amongst others) from Hunt et al. (2016) that are a factor 5 higher than what is currently used in work for the EC (Holland, 2014). Hunt et al. (2016) review the international literature on valuation studies of health impacts from air pollution, not only for bronchitis but also for hospital admissions, work loss days, restricted activity bronchitis in children.

6.4 Impacts on materials

In the absence of substantial new work on materials damage, and given the low contribution of this damage category to overall damage, it is appropriate to simply update previous cost data for repair or replacement of materials in line with inflation.

6.5 Impacts on crops and forests

Valuation of impacts on crops and forests has been performed (e.g. Holland et al., 2015b; Mills & Harmens, 2011; for a survey Castell & Le Thiec, 2016) using the assumption that pollutant damage is not sufficient to affect the price of either crops or timber on the world market, with world market prices preferred to national estimates to avoid national subsidies. For the specific application in quantification of damage linked to the emissions from specific industrial facilities this assumption seems reasonable. In any case, the use of more complex models for the economic evaluation of crop loss seems disproportionate when considering that associated impacts make up only a few percent at most of the total estimated damage.

Elements missing from the analysis include impacts from ozone induced crop loss on livestock and milk production, and interactions between air pollutants and other agents that stress plants, such as insect pests. In the absence of data for quantification of these effects it may be concluded that the damage costs are prone to underestimation. Analysis under the ECLAIRE study (Holland et al., 2015a) also showed that there are interactions between climate change and air pollution impacts on plants, in part through changes in the distribution of crop production (what species are grown where). However, any attempt to include such impacts would involve assumptions that again seem unlikely to improve the overall quality of the result.

An uncertainty in forest production concerns the extent to which changes in wood growth will feed through to a change in timber production. EEA (2017) shows that few European countries harvest at a rate that is equal to or in excess of wood production, and so it could be argued that a change in wood yield will not affect the trade in timber. It will, however, affect carbon sequestration by forests, which has been valued under ECLAIRE using a range for the marginal costs of GHG mitigation (the same range, €9.5 to 38.1/tonne CO_2 (2005 prices)[44] that was used by EEA (2014)).

6.6 Impacts on ecosystems and biodiversity

Ecosystem impacts were not included in the 2014 EEA assessment. Inclusion could be performed, for example using the approach from the ECLAIRE study as discussed above. We are unaware of new valuation data that can be applied to the change in critical loads exceedance beyond the studies of Christie on the UK's biodiversity action plan. If newer studies that provide marginal cost estimates (in other words, studies that value change rather than the total value of an ecological resource) are found it may be possible to introduce them to the analysis. However, on the basis of available results at the present time it is unlikely that this will add significantly to the damage estimates, as already noted.

[44] Corresponding to € 11.5 and 46.2 / tonne CO_2 (2016 prices).

Descriptive information on impacts to ecosystems (e.g. the extent of the problem of terrestrial eutrophication) should be provided to give context to the estimates for ecosystems. This information is already available from past analysis for the European Commission and the EEA. It is of concern that the rather small damage costs estimated for ecosystems are subject to very significant uncertainty against a background where there is widespread exceedance of critical loads.

6.7 Impacts from climate change or abatement costs for GHGs

As indicated in the previous chapter, from a welfare economics perspective the marginal abatement cost (or avoidance cost) approach is not a first-best solution as it does not directly measure and value all impacts of climate change. However, some argue that if the emission target adequately reflects the preferences of society it may be used to determine the society's willingness to pay for the related abatement level. The marginal abatement cost (MAC) approach can then be seen as a theoretically sound alternative. Ideally, to indicate the social desirability of the target, the targets should be confirmed by binding policies (EC, 2019; DEFRA, 2005; CE Delft, 2010). Others argue that adopting an abatement cost approach to evaluation would be to shift cost-benefit analysis very sharply in the direction of cost-effectiveness and priority-setting, rather than an assessment of overall net social benefits (Smith & Braathen, 2015).

The previously preferred approach for monetisation of GHG emissions in the EEA damage cost assessment is through the use of data on the marginal costs of GHG emission mitigation. A range was used in EEA (2014) of €9.5-38.1/tonne (2005 price, € 11.5 - € 46.2 in 2016 prices). It is noted that different values are being used in some Member States (see below).

The use of damage costs would increase coherence with the approach chosen for air pollutants. However, the values suggested in scientific publications or for use in policy assessment are prone to uncertainty, reflecting differences in possible scenarios of the future (economic growth, population growth, climate sensitivity, etc.) and the range of impacts considered (Auffhammer, 2018). Major assumptions also impact on estimates for abatement costs, such as the choice of target level, the assessment of mitigation options and their costs, the choice of baseline scenario… A recent review of both damage and avoidance cost value estimates published is presented in annex D of the Transport cost Handbook (EC, 2019) and shows the spread in the two types of values. Countries and institutions differ in their recommendations for the use of either damage or abatement costs. For policy appraisal, damage costs are favoured for example by the US, Canada and Germany, marginal abatement costs are favoured for example by the UK, France and the Netherlands. While the OECD argues in favour of the use of damage costs (OECD, 2019), the DG MOVE Transport Cost Handbook uses marginal abatement costs. We have not obtained clear guidance from DG CLIMA. The EEA has expressed a wish to choose an approach that is aligned to approaches supported by the different Commission Directorate Generals (DGs).

The German UBA in its Convention on methods for determining environmental costs (2018 & 2019), supports the use of damage costs for valuing GHG emissions. The damage cost factors to value CO_2 or other GHG emissions are indicated in Table 18, differentiated for three years and according to 2 different time-preference assumptions, represented by a social discount rate of 1% or 0%, respectively. The 0% is used for sensitivity analysis, as it gives the same importance to present and future benefits. For intermediate years, linear interpolation of the cost factors is recommended. The cost factors are based on the use of the damage cost model FUND (Version 3.0, Anthoff, 2007) and are thought to be situated amongst the lower range estimates of GHG damage costs. The recommended value of 180 €/t was also chosen because it gets close to the value of $€_{2016}$ 173,5 / t CO_2 recommended in IPCC (2014, page 691).

Table 18: UBA recommendations for climate damage costs, in € per tonne of CO_2 (€ price base 2016)

Tabelle 1: UBA-Empfehlung zu den Klimakosten in €$_{2016}$ / t $CO_{2\,äq}$

	Klimakosten in €$_{2016}$ / t $CO_{2\,äq}$		
	2016	**2030**	**2050**
1% reine Zeitpräferenzrate	180	205	240
0% reine Zeitpräferenzrate	640	670	730

Quelle: Eigene Darstellung.

Source: UBA (2019)

OECD (2019) presents the damage costs associated to a 2 °C scenario as estimated by Nordhaus (2014). In 2050 they converge with those recommended by the UBA (2019) but are lower for earlier years: 47.6 US$ in 2015, 94.4 US$ in 2030 and 216.4 US$ in 2050 (2005 prices)[45].

Climate change costs in the Handbook on the External Costs of Transport (EC, 2019) are based on the avoidance cost approach. The authors consider that damage costs have serious limitations because potentially catastrophic effects, such as the melting of the polar ice caps in Greenland or West Antarctica, cannot be well incorporated. Avoidance costs here are assessed as the costs necessary to reach the objective of the Paris agreement, i.e. to limit the global temperature rise to 1.5 – 2 °C. This implies a target of 80-95% reduction in CO_2 emissions by 2050 compared to 1990 levels. Based on a literature review the study concludes on the avoidance costs presented in Table 19 (details on the literature review are given in Annex D of the Handbook). These costs are the average of the values found in the literature, calculated for the minimum, maximum and central estimates. The preferred value for EC (2019) is the central estimate.

Table 19: Transport cost handbook proposal on climate change avoidance costs in €/t of CO_2 equivalent (€ price base 2016)

Table 24 - Climate change avoidance costs in €/tCO_2 equivalent (€$_{2016}$)

	Low	Central	High
Short-and-medium-run (up to 2030)	60	100	189
Long run (from 2040 to 2060)	156	269	498

Source: EC (2019)

The "Report on the High-Level Commission on Carbon Prices" (Stiglitz et al., 2017), suggests a minimum carbon-price level consistent with the Paris temperature target that is lower than the one suggested in EC (2019) but nevertheless in the same order of magnitude: at least US$40–80/tCO2 by 2020 and US$50–100/tCO2 by 2030.[46]

Values also in the same order of magnitude are used in the UK. UK guidance recommends figures (non-traded values) of £31-90/tonne CO2e in 2020, £35-106/tonne CO2e in 2030 and £100-302/tonne CO2e in 2050 (in 2010 prices, Department for Transport, 2019)[47].

In 2019, France has published its new guidance value used in evaluation of investment and policy decisions (Quinet, 2019). This value, based on the costs necessary to attain the policy objective of reaching carbon neutrality (or "zero net emissions", the residual gross emissions being absorbed by sinks)[48] in 2050, is set

[45] Approximately 53, 104 and 239 € in 2016 prices in 2015, 2030 and 2050, respectively.
[46] The price base for the dollar values is not indicated. Supposing it's 2016 prices the equivalent € values would be: at least €36–72/tCO_2 by 2020 and €45–90/tCO_2 by 2030.
[47] Equivalent to € 36-105/tonne in 2020, 41-124 in 2030 and 117-352 in 2050 (in 2016 prices).
[48] Assuming that if this objective was met by all countries, the 2 °C target could be met.

at € 250 (price base 2018, € 242 in 2016 prices) in 2030 and in a range from € 600 to €900 € in 2050 (mean €775)[49].

For valuing CO2, CE Delft (2018) suggests values based on the marginal abatement cost approach. Values are suggested for two scenarios, a business as usual scenario, and a scenario expected to respect the 2°C target (as formulated as objective under the Paris agreement). The latter are not very different from those suggested by EC (2019). The values are presented in the following table (Table 20).

Table 20: CE Delft (2018) proposal for CO2 costs in €/t of CO_2 equivalent (€ price base 2016) based on a marginal abatement cost approach

	2015	2020	2030	2040	2050
Current policy	48	57	80	113	160
2°C policy	80	95	130	180	261

Given the quantity of research on climate and climate mitigation at the present time, further review of the literature could be undertaken to identify new estimates of damage or new approaches to valuation of greenhouse gas emissions. Comparison could be made also with more alternative figures for marginal abatement cost (MAC) values, used in other Member States. However, given the potential for variation in estimates of future climate damage and climate change mitigation, it is considered here that it is unlikely that such research would provide a firmer basis for quantification than currently used.

The following table (Table 21) summarises the discussed values.

[49] In 2016 prices the values are €581, €871 and €750, respectively.

Table 21: Summary table on carbon prices (in €, price base 2016)

Low estimate	Central estimate	High estimate	Year	Type of value	Scenario context	Region	Source
12		46	2012	MAC		EU27	EEA 2014
60	100	189	up to 2030	MAC	2°C scenario	EU28	EC 2019 (DG MOVE)
156	269	498	2040 to 2060	MAC	2°C scenario	EU28	EC 2019 (DG MOVE)
36		72	2020	MAC	2°C scenario	global	Stiglitz et al. 2017
45		90	2030	MAC	2°C scenario	global	Stiglitz et al. 2017[50]
36		105	2020		guidance for non traded values	UK	DFT 2019
41		124	2030		guidance for non traded values	UK	DFT 2019
117		352	2050		guidance for non traded values	UK	DFT 2019
	242		2030	MAC	2°C scenario	France	Quinet 2019
581	750	871	2050	MAC	2°C scenario	France	Quinet 2019
	53		2015	MAC	2°C scenario		OECD 2019, Nordhaus 2014
	104		2030	MAC	2°C scenario		OECD 2019, Nordhaus 2014
	239		2050	MAC	2°C scenario		OECD 2019, Nordhaus 2014
	80		2015	MAC	2°C scenario	EU28	CE Delft 2018
	130		2030	MAC	2°C scenario	EU28	CE Delft 2018
	261		2050	MAC	2°C scenario	EU28	CE Delft 2018
180		640	2016	damage cost	low (high) value 1% (0%) time preference rate	Germany	UBA 2019
205		670	2030	damage cost	low (high) value 1% (0%) time preference rate	Germany	UBA 2019
240		730	2050	damage cost	low (high) value 1% (0%) time preference rate	Germany	UBA 2019
	174		2013	damage cost	Average across various studies (1% pure rate of time preference)	global	IPCC 2014

[50] The price base in which the original values are expressed in this study is not specified. We assume the price base was 2016.

From a scientific point of view, we would favour the use of damage costs for CO_2. However, given the EEA's preference for approaches that can be supported by other DGs, the absence of DG CLIMA guidance on damage costs and the fact that DG MOVE via its transport cost handbook supports a marginal abatement cost approach, direct valuation based on estimates of costs necessary to reach the Paris agreement seems the most pragmatic approach in the short term. Communication with CE Delft and Trinomics involved in the ongoing studies for DG ENV and DG ENER (see above) suggest they are likely to follow the approach used in the DG MOVE Transport Cost Handbook. Comments received from DG ENV underlined the importance they give to consistency with DG MOVE. The values to be used should ideally be accepted by DG CLIMA.

Conclusions from the pollutant screening (chapter 3.4) suggest methane and N_2O as GHGs may also be included in the assessment. In general, given the simplicity of using damage per tonne estimates to an aggregate of GHG emissions weighted by their global warming potentials, other figures can be factored into the analysis without trouble. It should be noted however, that estimates of avoidance costs for CO_2 may not be proportional to that of avoiding methane or N_2O (EC, 2019).

6.8 Updating of the Euro price base

It is assumed that the price year for the analysis should be 2019 or 2020 for new analysis. Policy studies for the Commission still use 2005 prices for the benefits assessment, for consistency with the cost data for abatement options included in the GAINS model. This adjustment can be made using standard factors available from Eurostat.

In previous EEA assessments, unit costs per impact endpoint quantified were updated for price base but not for income growth. Ricardo (2019) applies income growth updates relative to the year for which costs were first assessed, using observed real income growth rates for past years and an uplift factor of 2% for the future (the latter corresponding to Department of Health guidance). UBA (2019) supports increasing willingness to pay (WTP) estimates with income. The adaptation includes furthermore a hypothesis on the income elasticity. For the primary aim of the current research, providing estimates of damage costs and applying them to emissions in a previous year it is not necessary to discount, or to inflate estimates to a different price base. However, the damage cost data are widely used outside of this work (e.g. in assessment of requests for derogation under the Industrial Emissions Directive), and as such, guidance on the application of the data should be provided. It is recommended that this guidance should include both discounting at 4% and inflating future damage estimates by 2% per year.

6.9 Conclusions taking account of feedback from the expert consultation

For the classical air pollutants, cost data for morbidity impacts used in the previous assessment is rather dated and newer information should be considered (e.g. more recent data provided in Ricardo, 2019 or Andersen et al., 2019). However, any revision of the figures should be carried out in agreement with European Commission services to ensure that there are not inconsistencies in approach across the Commission and its agencies. This position was agreed with by the majority of experts consulted. We suggest presenting in the 2020 report recently updated morbidity costs (e.g. Andersen et al., 2019) for comparison with the applied values.

Cost data for mortality have been updated in the recent DG MOVE Transport cost handbook, basing their estimate on the OECD (2012) value for a VSL. We suggest doing the same here. Agreement with Commission DGs is of course important to ensure coherence across different assessments. Note that

the European Commission "Better Regulation Toolbox" both of 2015[51] and of 2017[52] makes reference to the same OECD value.

The issue of whether to quantify mortality based on premature deaths (monetised by VSL), on years of life lost (monetised by VOLY), or on both, remains unresolved amongst scientists. This ongoing dispute was also reflected in the feedback we received during our expert consultation. It ranged from the recommendation to use solely VSL, through using VSL for acute and VOLY for chronic mortality, to using VSL and VOLY as alternative values given that they answer slightly different questions. We suggest continuing using both values in the upcoming EEA assessment and to present them as two alternative approaches.

Concerning the valuation of mortality, there are several questions:

1. **Whether to use the VSL or VOLY?** Given divergent views in Member States and amongst the experts consulted here, it remains appropriate to use both.
2. **What methods are appropriate to calculation of the VSL and VOLY?** For the VSL, there is the clear recommendation of OECD (2012) to adopt estimates based on willingness to pay approaches (WTP). For the VOLY, practice has been split between estimates derived from original WTP research (e.g. Desaigues et al., 2012) and estimates derived from the VSL, treating assumed remaining life expectancy as a discounted stream of annual values. The latter approach was first discussed in the context of environmental externalities by ExternE (1995, 1998), drawing partly on earlier work by Harrison (1990). Discussions at the time regarded it as a stop-gap approach, prior to original valuation work on the VOLY. It is notable that rather little research of this type has been carried out in the 25 years since the first ExternE report was published, but also that the VOLY generated in ExternE (70kECU in 1990 prices, estimated here to be equivalent to €117 in 2016) are of a similar order of magnitude to those in use today, especially for the mean estimates of VOLY.
3. **Whether to adopt the mean or median values of the VSL and VOLY.** This question has received surprisingly little discussion over the years since it was first adopted under the CAFE (Clean Air For Europe) programme in the early 2000s, although it makes a factor 2 difference to the mortality results. Whilst it has been argued that the median is more democratic, it downplays the opinions of those with high valuations.
4. **What studies should values be based on?** For the VSL there is only one logical choice, and that is the 2012 OECD study, as it provides a meta-analysis of all relevant contingent valuation studies. It is understood that the 2012 report is currently being updated: it would be logical to follow the conclusion of the updated research when it is available, though it may be little different to the existing conclusions, given that the new work will inevitably account for much of the literature covered in the first report. The literature on VOLY is far more limited than it is for VSL. Whatever value is adopted for the VOLY should be checked for consistency with the adopted VSL estimate.
5. **Should country-specific values be adopted?** The previous damage costs from EEA (2014) used EU-averaged costs. It is proposed that this is retained for further work. There are some situations where country-specific values may be preferred. If this is the case, conversion of the EU-averaged data can be made using guidance from OECD's meta-analysis[53].

[51] https://ec.europa.eu/info/sites/info/files/better-regulation-toolbox-2015_0.pdf

[52] http://www.emcdda.europa.eu/system/files/attachments/7908/better-regulation-toolbox_1.pdf

[53] http://stats.oecd.org/Index.aspx?DataSetCode=EXP_MORSC provides harmonised country-specific and EU-wide VSL estimates from 1990 to 2017.

Further to this discussion, it is noted that the lower bound VOLY (median) used in the previous EEA (2014) assessment is equal (after adjustment from 2005 prices to 2016 prices) to the VOLY used in the DG MOVE transport handbook (70 k€), which is likely to be used also in the ongoing DG ENV study "Mapping objectives in the field of environmental taxation and budgetary reform: Internalisation of environmental external costs" (personal communication with CE Delft). Further personal communication with Trinomics indicates that the values considered for the ongoing DG ENER study on energy costs, taxes, subsidies and external costs also draw heavily on the values used in the DG MOVE study and the Environmental Handbook (CE Delft, 2018). A final conclusion needs to take account of the issues raised above.

In the absence of substantial new work on materials damage and given the low contribution of this damage category to overall damage, we suggest updating previous cost data for repair or replacement of materials in line with inflation.

It could be argued that damage to agriculture from ozone might be underestimated: impacts not yet accounted for comprise impacts on livestock and milk production, and interactions between air pollutants and other agents that stress plants. Also, previous EEA and CLRTAP assessments only covered a limited number of crop species. The formulation of the ozone dose as given in the CLRTAP Manual (2017) is tending towards a high estimate. Whether this might carry through to estimated impacts would require more research. The ECLAIRE (2015) study did cover all crops.

Whereas the Mapping Manual (LRTAP Convention, 2017) has reservations about valuation of ozone impacts on forest for uncertainty reasons, the ECLAIRE study team involving various scientists working under the LRTAP Convention decided that uncertainty would be higher if leaving out forest valuation. We follow this view. We do not suggest any change to the valuation of crop and forest damage. The simple approach of using producer prices, country specific or European average, appears justifiable given the limited contribution of this endpoint to the overall damage costs.

Effects on ecosystems and biodiversity were not accounted for in the 2014 assessment. New assessments are available but are limited in number. Estimated economic damage remains small compared to health damage from air pollution. There is a possibility that willingness to pay studies lead to a bias towards underestimation of damage, possibly because survey respondents might consider that government should carry a large share of the responsibility for ensuring that ecosystems are protected. This issue needs more investigation. Uncertainty is high also with respect to the quantification of biodiversity impacts. Nevertheless, given the political and public interest in biodiversity losses, we suggested in chapter 5.5 to include such effects into the assessment. We suggest basing the assessment in 2020 on WTP values[54].

Some recent studies provide valuation data for health endpoints that were not previously included for metals (Nedellec and Rabl, 2016a, b; Ricardo, 2019). If the new endpoints are to be included in the analysis, we would suggest basing their valuation on these studies.[55] The use of the cited studies would also profit from agreement with European Commission services. Issues surrounding the potential for double-counting with assessment of $PM_{2.5}$ (given that most of these pollutants tend to be closely associated with PM emissions) were discussed above.

[54] The use of restoration costs or estimates of the loss of ecosystem services are alternative approaches.

[55] However, whether the inclusion of new impacts from metals is feasible in the short term will be studied in 2020 (cf. chapter 5.5).

Although the use of damage costs for carbon valuation would be more in line with the approach applied to the valuation of health effects and present the scientifically sound approach (and most consulted experts expressed this opinion), we are not aware of any Commission level guidance on the use of damage cost estimates for CO_2. In the absence of DG CLIMA guidance on damage costs and given that DG MOVE via its Transport Cost Handbook promotes a marginal abatement cost approach, direct valuation would seem the most pragmatic approach to valuation. Communication with CE Delft and Trinomics involved in the ongoing studies for DG ENV and DG ENER suggest they are likely to follow the approach used in the DG MOVE Handbook. Chosen values should ideally be accepted by DG CLIMA. In the absence of further guidance from the EC we suggest following the approach and values applied in the DG MOVE study.

Furthermore, following conclusions from chapter 3, methane and N_2O as greenhouse gases should be incorporated into a measure of CO_2 equivalents. A caveat, however, is that estimates of avoidance costs for CO_2 may not be proportional to that of avoiding methane or N_2O.

In this report we have taken the view that uncertainty of damage cost estimates is reduced when adding in more factors for which reasonable evidence is available than when leaving them out. We suggested excluding factors mainly when their impact is low and when the effort to include a detailed assessment would draw a large resource away from effects that have been shown repeatedly to be far more important.

One might engage a more general discussion on how to deal with individually small contributions (such as ecosystems impacts, impacts on timber production, other non-health ozone impacts not yet included (livestock production, materials ...)) that might be significant if added together. Our recommendation is to factor in as many effects as possible, while highlighting uncertainty. However, alternatives for dealing with non-dominant impact factors could be either to excluded them altogether from cost estimates, or to estimate a lump-sum addition (WHO/OECD, 2015, for example, estimates morbidity from PM2.5 as equal to 10% of the pollutants mortality damage; Hunt et al. (2016) estimate it at 10% to 15% of mortality costs).

The majority of the experts consulted agreed that from an economic perspective, when reasonable evidence on a damage cost estimate exists, it should be included in the analysis, as not doing so means setting the corresponding cost to zero. Experts further agreed that too much effort should not go into exploring effects that have been proved to be very low, but when robust estimates are available and easy to compute, these effects should be included. We therefore suggest adding in factors for which reasonable evidence is available, and not to rely on lump-sum additions.

7 Conclusions

This working paper reviews the previous EEA assessments of damage cost per tonne of atmospheric pollutant and identifies areas in which scientific developments make changes to the previous methods feasible. It takes into consideration feedback received from experts during a written consultation on an earlier draft that took place in autumn 2019. In its structure the paper follows the steps of the impact pathway approach. It should be noted that both EEA and EC representatives have expressed the wish for the choices suggested here to be as consistent as possible with EC guidance and recent or ongoing EC studies.

On the ***emission*** side, an analysis of frequency of reporting in the E-PRTR database and quantity of reported emissions led to the recommendations

a) to continue covering the pollutants assessed in earlier reports, i.e.
   - ammonia (NH_3), nitrogen oxides (NOx), non-methane volatile organic compounds (NMVOCs), particulate matter ($PM_{2.5}$, PM_{10}) and sulphur oxides (SO_2),
   - arsenic, cadmium, chromium, lead, mercury and nickel,
   - benzene, dioxins/furans and polycyclic aromatic hydrocarbon.
b) to additionally take into consideration methane and nitrous oxide, and carbon monoxide, copper, zinc, fluorine and chlorine.

An additional screening of ecotoxicity and human toxicity suggested vinyl chloride, dichloromethane and 1,2-dichloroethane might be worth including in the analysis as well.

Concerning ***dispersion and exposure modelling*** for the major air pollutants, we recommend the use of the most recent and validated EMEP Source Receptor Matrices (SRMs). Concerning the impact of ozone on agriculture, AOT40 SRMs will need to be used as long as PODy SRMs are not available.

For the assessment of NO_2 health impacts, the use of the CHIMERE-SHERPA model with 7km×7km grid-to-grid SRRs is feasible, with estimated errors associated to the grid resolution below 20% for no-threshold indicators. The current resolution of the CHIMERE-SHERPA tool (7km) still smoothens out NO_2 gradients, making it unfit to estimate population exposure to NO_2 concentration levels above a threshold (such as the 20 $\mu g.m^{-3}$ cut-off point as in the response function currently recommended by the WHO/Europe). A more spatially resolved CTM-SHERPA would be required to address this issue. Our suggestion is to use CHIMERE-SHERPA (and/or EMEP-SHERPA if available) with its current resolution for the 2020 update for response functions without cut-off point.

The desirability of deriving a unit damage costs (€/tonne of pollutant) specific to the industrial sector was identified in the EEA (2014) report and confirmed by the consulted experts. However, it is apparent that there is a lack of consensus on the best approach to dealing with this sector specificity. The results of the earlier EURODELTA II study are now outdated, so can no longer be regarded as state-of-the-art. Assessment carried out as part of this work finds that there are three options for proceeding:

1. Adopt average damage costs by country that do not take account of the specificity of the industrial sector. Given that the observed potency ratings from analysis so far using the SHERPA models are generally close to 1 (i.e. close to the national average damage cost), this position, though imperfect, may be a pragmatic solution to the problem.
2. Use the results of a specific model (SHERPA-EMEP or SHERPA-CHIMERE).
3. Use results averaged across SHERPA-EMEP and SHERPA-CHIMERE.

At the present time, limited results are available from the SHERPA models, especially regarding the temporal representativeness since only one year is targeted in the parametrisations. Further modelling is needed to better understand the way that the results for different sectors vary across Europe. In

particular, countries distant from the centre of Europe (e.g. Greece, Portugal, Finland for which the analysis has not yet been carried out) should be added to the analysis prior to making a final decision on the most appropriate approach and source for sector adjustments. Inclusion of these countries will improve understanding of the extent and reasons for variation. This work should be finalised by the end of Q1/2020 before the new methodology needs to be implemented.

Concerning metals and other pollutants that can cause damage through a variety of pathways covering ingestion as well as inhalation, there is variation in modelling frameworks, between simplified tools as used in EEA (2014) and more complex systems such as Pangea (Wannaz et al., 2018a, b, c). The consistency of the Pangea framework with existing European tools (EMEP, CHIMERE, etc.) for air pollution assessment is not known in detail. Consistency with other tools, however, seems good (e.g. EUSES, given the links with USEtox). It is also noted that complex models may not give clearly better results than simpler tools.

There is a balance to be struck between complexity, transparency and flexibility. Without systematic testing of the different tools it is not clear which are best suited to the current application. Perceived increase in accuracy of the exposure assessment for toxic metals and specific organic pollutants of relevance to the work under discussion needs to be seen in the wider context of the final estimates of damage per unit emission. Discussion in this working paper highlights greater uncertainties at the stage of defining which impacts should be quantified than seem likely to originate from the exposure assessment. We therefore conclude that exposure methods do not need to be updated compared to the 2014 assessment, while more focus should be given to the impact assessment. This position may be reviewed in the future, for example if the list of pollutants requiring quantification was to be extended.

With respect to the ***quantification of impacts*** from air pollution we suggest continuing providing country specific unit damage costs for emissions of the major air pollutants that cover the impact of emission per tonne of pollutant from a particular country, wherever the impacts occur. We suggest to additionally present the damage occurring only in the country.

There is some interest (for example from OECD) in splitting out the costs of air pollution to business (especially through impacts on productivity). These have always been included in the damage cost analysis for the EEA, though the methods used to date are to be reviewed and possibly updated by the end of February 2020. This active field of research will be kept under review up to the time that the models are run.

For additional health effects included in recent analyses, but not in previous EEA assessments (such as cardio vascular disease or stroke), we will conduct a sensitivity assessment in 2020 but we consider inclusion in the best estimates at the present time as premature.

There are a few publications suggesting updated (non-linear) response functions for mortality effects from $PM_{2.5}$. In the absence of a review of response functions by the WHO, we suggest continuing using the HRAPIE (WHO, 2013a) response functions for all-cause mortality. The possibility of a side evaluation (sensitivity calculation) with a more recent response function will be considered in the analysis in 2020.

Health impacts from NO_2 were not included in the previous EEA assessment. We suggest extending future EEA assessments to include health impacts from NO_2. Morbidity impacts from NO_2 relying on response functions without cut-off points can be accounted for in the damage cost assessment. Concerning chronic mortality, however, different exposure-response functions are currently used. Applying the response function with a cut-off point recommended by HRAPIE together with the available source-receptor matrices would involve high uncertainty and would tend to underestimate mortality effects from NO_2. Therefore, for the 2020 analysis we suggest the following. If the WHO proposes a new response function without a cut-off point, we will use this function. Otherwise, we will not include NO_2 chronic mortality into this year's damage cost assessment. However, a screening

analysis to assess sensitivity of results to different available response functions for NO_2 chronic mortality on overall health damage will be carried out based on high resolution chemistry transport modelling in France. In this screening we will compare the HRAPIE function with the different COMEAP (2018) views.

We suggest extending the assessment of the impacts of air pollution on materials in future analyses and to include, on top of corrosion effects on materials, also impacts of soiling from particulate matter and impacts of ozone. Furthermore, information on updated response functions for materials that may have changed since the currently used functions were developed (e.g. paints) should be collected in the 2020 assessment.

For trace metals and organic pollutants, recent analyses assess impacts not previously included in EEA assessments (e.g. mortality). Given, however, that these assessments are made for the birth cohort rather than for the population alive as is the case for other health impacts assessed in earlier EEA reports, plus the need to pay attention to the risk of double counting, for example when chronic mortality not only of PM but also of mercury and lead is taken into account, we conclude that further assessment is required before additional impacts of metals can be included in the analysis.

The scientifically recommended indicator to assess impacts on crops and forests from ozone is the stomatal ozone flux. However, currently no PODy SRMs are available. For the time being, the calculation of impacts of ozone on crops and forests will hence need to continue using the AOT40 indicator as in the earlier EEA (2014) report. For the future we recommend the creation and publication of POD SRMs.

Although uncertainty in quantifying ecosystems and biodiversity impacts is still high, we suggest calculating biodiversity effects from deposition of NH_3 and NO_2 for exceedances of critical loads in Natura 2000 areas. While monetised impacts are likely to remain low, the political importance of biodiversity, and the extent of critical loads exceedances for nitrogen, are high.

With respect to pollutants whose integration was a suggestion from the screening analysis in Chapter 3, we are not aware of broadly agreed dose-response functions. Out of the 8 pollutants (carbon monoxide, copper, zinc, fluorine, chlorine, vinyl chloride, dichloromethane and 1,2-dichloroethane) and 2 greenhouse gases (methane and nitrous oxide) whose integration was suggested from the screening analysis, we suggest retaining only the two greenhouse gases. The contribution of methane to ozone, however, cannot be accounted for as there are currently no SRMs for methane.

The consulted experts agreed with us that biomass use is not necessarily carbon neutral. The scientific debate on what assumptions to use with respect to the CO2 emissions of biomass combustion is unresolved. We suggest applying a somewhat arbitrary middle option between assuming 0 and 100% carbon emissions.

The final issue reviewed in this working paper is the ***valuation*** data available. Although cost data for morbidity impacts used in the 2014 assessment are rather dated, we suggest that any revision of the figures should be carried out in agreement with European Commission services to ensure that there are not inconsistencies in approach across the Commission and its agencies. In the 2020 report we will present recently updated morbidity costs for comparison with the applied values.

The issue of whether to quantify mortality based on premature deaths (monetised by VSL), on years of life lost (monetised by VOLY), or on both, remains unresolved amongst scientists. This ongoing dispute was also reflected in the feedback we received during our expert consultation. We suggest continuing using both values in the upcoming EEA assessment and to present them as two alternative approaches.

Cost data for mortality have been updated in the recent DG MOVE Transport cost handbook, basing their estimate on the OECD (2012) value for a VSL. The European Commission "Better Regulation

Toolbox" both of 2015 and of 2017 makes reference to the same OECD value. We therefore suggest applying the same VSL estimate also in the 2020 EEA assessment.

The DG MOVE transport handbook applies a VOLY of 70 k€. According to personal communication, it is likely to be used also in the ongoing DG ENV study "Mapping objectives in the field of environmental taxation and budgetary reform: Internalisation of environmental external costs". Whatever value is adopted for the VOLY should be checked for consistency with the adopted VSL estimate.

There has been no substantial new work on materials damage. We suggest updating previous cost data for repair or replacement of materials in line with inflation.

With respect to valuing damage from ozone on crops and forests we do not suggest any change to the valuation. The simple approach of using producer prices, country specific or European average, appears justifiable given the limited contribution of this endpoint to the overall damage costs.

Effects on ecosystems and biodiversity were not accounted for in the 2014 assessment. New assessments are available. Although estimated economic damage remains small compared to health damage from air pollution, given the political and public interest in biodiversity losses, we suggested to include such effects into the assessment. We suggest basing the assessment in 2020 on WTP values.

Concerning carbon valuation, the use of damage costs for carbon valuation would be more in line with the approach applied to the valuation of health effects and present the scientifically sound approach. Review of current approaches shows that most EU countries that publish guidance apply the marginal abatement cost approach. An exception is Germany. We are not aware of any harmonised EC guidance on carbon valuation. Nevertheless, DG MOVE in its transport cost handbook promotes a marginal abatement cost approach. Communication with CE Delft and Trinomics involved in the ongoing studies for DG ENV and DG ENER suggest they are likely to follow the approach used in the DG MOVE Transport Cost Handbook. Chosen values should ideally be accepted by DG CLIMA. In the absence of further guidance from the EC we suggest following the approach and values applied in the DG MOVE study.

In the course of this paper a number of possible ***sensitivities*** have been identified. Inclusion of all of these sensitivities in the final analysis is unlikely to be necessary, and would make the assessment much more difficult to follow. Analysis will demonstrate which of these are most important and should be retained.

Concerning ***country scope***, most countries included in the E-PRTR can be covered by the analysis. The assessment will include the 27 EU Member States, the UK, Norway, Iceland and Switzerland. Serbia continues to be an exception, because of the lack of reported CO_2 data.

Further on the issue of ***consistency*** between current studies for the European Commission and the EEA, it might be useful to consider comparability beyond the impact indicators and unit values applied in the assessments. While the work in the DG MOVE Transport Cost Handbook estimates background pollutant concentrations on the basis of the relationship between damage and emissions for various emission scenarios from NEEDS (2008a), the 2020 EEA study will be based on up to date pollution information, using recent EMEP SRMs.

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Annex 1
Sectoral Emission Potencies

Relying on a unique national average estimate of damage per tonne of pollutant ignores the proximity of industry with population and in some cases the release of pollutants at high altitude that can introduce strong differences in exposure and health impact. One way to reduce associated errors is to differentiate the national damage per tonne of pollutant by activity sector. This sectorization has been added to the EEA (2014) methodology using results from the first report on sectoral approaches from the Eurodelta-II exercise (Thunis et al., 2008). In that report, sectoral potencies were elaborated to reflect the relative efficiency of a specific sector compared to all sectors in reducing $PM_{2.5}$ exposure by reducing precursor emissions. In the EEA 2014 report, these sectoral potencies were referred to as "correction factor", but the term potency is preferred here for clarity.

Unfortunately, there has not been any update of the Eurodelta-II exercise. And already at the time of the 2014 report, the limitation that those results were only relevant for a handful of European countries was brought forward. In order to explore the feasibility of updating these potencies, the SHERPA model has been used. This annex presents the methodology used to calculate the potencies with the SHERPA tool, an analysis of the results for 6 countries and the comparison with previous potencies from Eurodelta-II.

A1.1 Methodology

The methodology used to estimate country-based and sectorial potencies is similar to the one developed in Eurodelta-II and used in the EEA 2014 report. The main difference is that we do not rely on a full-CTM but on the SHERPA surrogate model that provides the grid-to-grid impact of emission reductions (NOx, NH_3, PPM, SO_2, NMVOC) to $PM_{2.5}$ and NO_2 concentrations. To reproduce CTM concentrations consecutive to emission reductions, SHERPA needs to be trained based on scenario simulation of the full CTM. Two versions of the SHERPA tool exist. One is based on the CTM CHIMERE (Menut et al., 2014) run at 7km horizontal resolution for the meteorological year 2009, with emissions based on GAINS total emissions per country-pollutant-sector for 2010, gridded with proxies from the MACC-TNO emission inventory from the year 2010 and a specific proxy for residential heating sector (Terrenoire et al., 2015). Specific national inventories for France and UK are also used in CHIMERE-SHERPA (Thunis et al., 2016). Meteorological conditions are extracted from IFS for the year 2009. This version is available online on the SHERPA webpage[56] . Another version has been recently developed (Pisoni et al., 2019) based on the EMEP MSC-W model 4.9, run for meteorological conditions from 2014, with a resolution of 0.1° and with emissions provided by JRC for the year 2014 (Trombetti et al., 2017). Both versions of the SHERPA tool (CHIMERE-SHERPA and EMEP-SHERPA) have been used to estimate the sectoral potencies.

The main steps followed to calculate the potencies are summarized here:

1) Firstly, an emission efficiency is calculated with respect to the population exposure over the EU-28 countries. This efficiency is a measure of the response in terms of population exposure (i.e. pollutant concentration multiplied by population) to an emission reduction. To be comparable from a sector to another, efficiency is normalized by the difference in emissions:

[56] https://ec.europa.eu/jrc/en/news/sherpa-computational-model-better-air-quality-urban-areas

$$\text{efficiency}=\frac{\sum_i \Delta Concentration_i \times Population_i}{\sum_j \Delta Emission}$$

with i=grid points over EU-28 countries

j=grid points over the country

ΔEmission: difference in net emission (kg) corresponding to a 15% decrease. ΔConcentration=resulting reduction in concentrations (µg/m3)

This emission efficiency (here expressed in $\mu g.m^{-3}.kg^{-1}$) is calculated for each sector independently and for the case where emissions of all activity sectors are reduced at the same time (by the same percentage). The higher the efficiency, the higher is the potential to reduce exposure by a reduction in 1 kg of emitted pollutant.

2) The relative efficiency of a specific sector compared to the efficiency of "all sectors" is evaluated through the calculation of the potency for sector S:

$$\text{potency} = \frac{Sector\ S\ efficiency}{Allsector\ efficiency}$$

A potency > 1 means that emission reductions focused on this sector are more efficient than a homogeneous reduction of emissions over all sectors.

Potencies can be calculated for all individual activity sectors, with a particular interest in industrial activities in the context of this report. Using the SNAP terminology, we included the following macro-sectors:

- SNAP 1 sector: combustion in energy and transformation industries, mainly thermal power stations and urban heating, from large sources (> 300 MW) to smaller ones (<50 MW). Emissions emitted from tall stacks (high level sources). Sources not uniformly distributed across the country.
- SNAP 3 sector: combustion in manufacturing industries, from large sources with tall stacks to smaller ones with emissions released at low levels. Sources distributed over the whole country, mainly in industrial areas.
- SNAP 4 sector: Production processes. Mainly low-level sources distributed over the whole country, mainly in industrial areas.
- SNAP 9 sector: Waste treatment and disposal. Mostly high-level sources.

A1.2 Spatial correlation between emission location and population density

Before describing results obtained with the SHERPA surrogate model, it is relevant to explore the spatial correlation between emissions and population over the country, depending on the sector under consideration. High spatial correlation simply indicates a co-localisation between emissions and population without considering the level of emission or size of population. These correlations reflect the emission inventory used in the CHIMERE model.

In a first approximation, sectors showing the largest spatial correlation between emissions and population should also show potencies higher than one. Figure 11 and Figure 12 show the spatial correlation between emissions and population over each country for NO_x and PPM, respectively. In addition to the industrial SNAPs described above (SNAP 1, 3, 4 and 9), correlations were also calculated for SNAP 2 (combustion in non-industrial plants, mainly residential-service sectors), SNAP 6 (solvent use), SNAP 7 (road transport), SNAP 8 (off-road transport) and SNAP 10 (agriculture).

Figure 11: Correlation (r) between NO_x emissions (as used in SHERPA-CHIMERE) and population over each grid of the targeted country

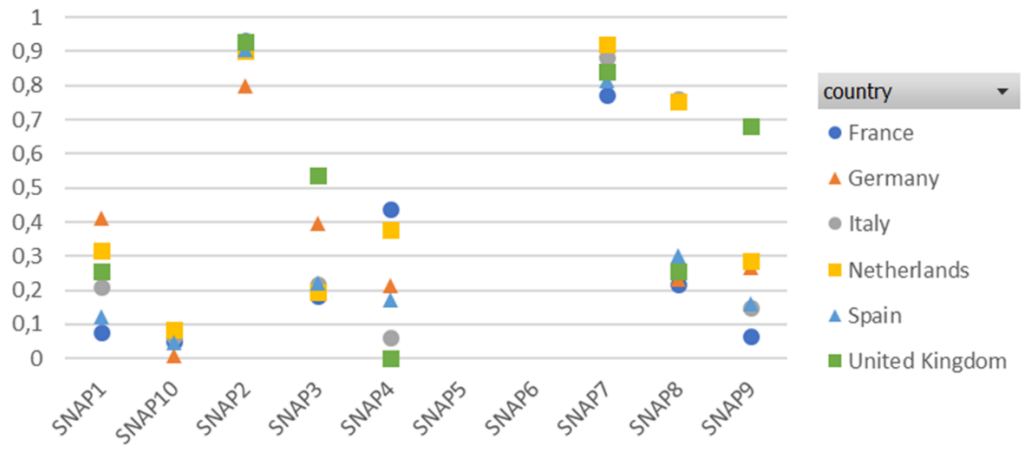
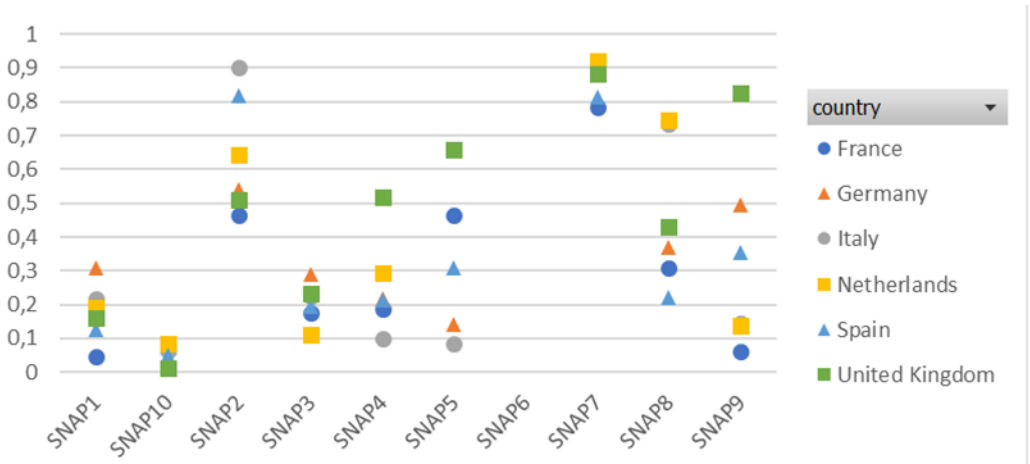


Figure 12 : Correlation (r) between PPM emissions (from CHIMERE-SHERPA) and population over each grid of the targeted country



- The lowest correlations are logically found for the agriculture sector (SNAP 10) followed by the SNAP1 sector. Fields and farms are generally located in rural areas, and emissions for the SNAP 1 sector come mainly from large point sources disparately distributed over the country.
- The highest correlations are generally found for the residential sector (SNAP 2) and the road sector (SNAP 7). However, it should be noted that correlation between PPM and population for SNAP2 is not as high as expected given that the inventory is based on a population proxy for disaggregating the SNAP2 national value. This is due to an additional parametrisation in the inventory to take into account the fact that there is an increase in the relative contribution of wood burning in the fuel mix when moving from urban centers to rural areas (e.g. due to an increase in domestic fireplaces). A logarithmic regression function is added to estimate PPM from SNAP2 with emissions per inhabitant sharply decreasing when the population density increases. This effect is noticeable only for $PPM_{2.5}$, because biomass burning sources contribute to a very large fraction of PM emissions in SNAP2, while they emit a much lower fraction of gas phase pollutants.

- For some sectors, a large disparity in correlations is found between countries, such as for the waste sector (SNAP 9) for which correlation values range from 0.06 (France) to 0.82 (UK). This is generally the case for all industrial sectors. Emissions from these sectors partly come from large industrial sites that report emissions to E-PRTR and that are included in the emission inventory as point sources. Their proximity to the population will then vary between countries. For the residential sector, part of the national emissions is spatialized using a population proxy, explaining a larger correlation with population.

Based on the analysis of the correlation between emissions and population, it could be expected as a first approximation that potencies calculated for the SNAP 1 sector, showing a low correlation between emissions location and population density, will be lower than one. This is not the case for all countries, as shown in section A1.2. In fact, the degree of spatial correlation between emission and population does not reflect information on the quantity of emissions. A low correlation can be found if most of the emitters are located far from population, but if one emitter accounting for the majority of the emissions is located in a very populated area, then the total potency will be higher than 1. In order to investigate this point, an emission indicator was constructed to reflect both the quantity of pollutants emitted and the spatial correlation between emissions and population.

1) population weighted emissions normalized by emissions were computed for each pollutant, sector and country as follows:

$$\text{Emission-PopW}=\frac{\sum_i Emission_i \times Population_i}{\sum_i Population_i \times \sum_i Emission_i};$$

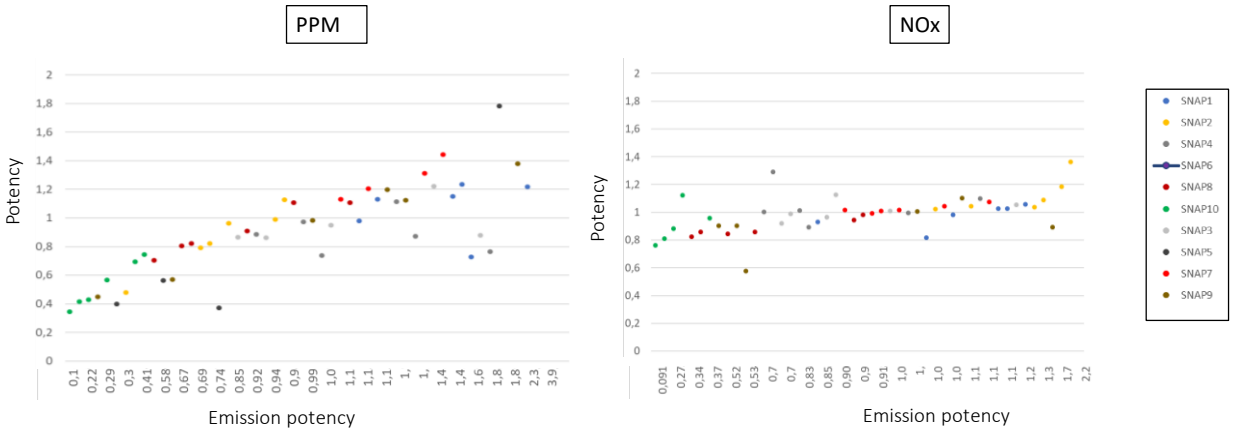
2) based on the same equation as used to calculate the exposure potency described in section A1.1, an "emission potency" ratio has been constructed as an indicator of the sectoral proximity of emissions to population:

$$\text{Emission-potency}=\frac{(Emission\ PopW)_{sector}}{(Emission\ PopW)_{all\ sector}}$$

An emission potency > 1 means that, when weighted by population, the normalized emissions of the targeted sector are higher than the normalized emissions of all sectors. This can be the case for emissions well correlated with population (i.e. road transport – SNAP7) but also for sectors less corelated with population at the national level but exhibiting high spots of emissions close to dense areas (i.e. SNAP 3 for some countries and pollutants). This emission potency depends only on the emission inventory, whereas potencies (based on concentrations) are calculated on the CHIMERE-SHERPA response.

Emission potencies and potencies calculated with the CHIMERE-SHERPA surrogate model are compared in Figure 14 for all sectors, all pollutants and all countries.

Figure 13: CHIMERE-Sherpa based potencies versus Emissions-potencies ratio for PPM (left) and NOx (right) in the emission inventory used in SHERPA

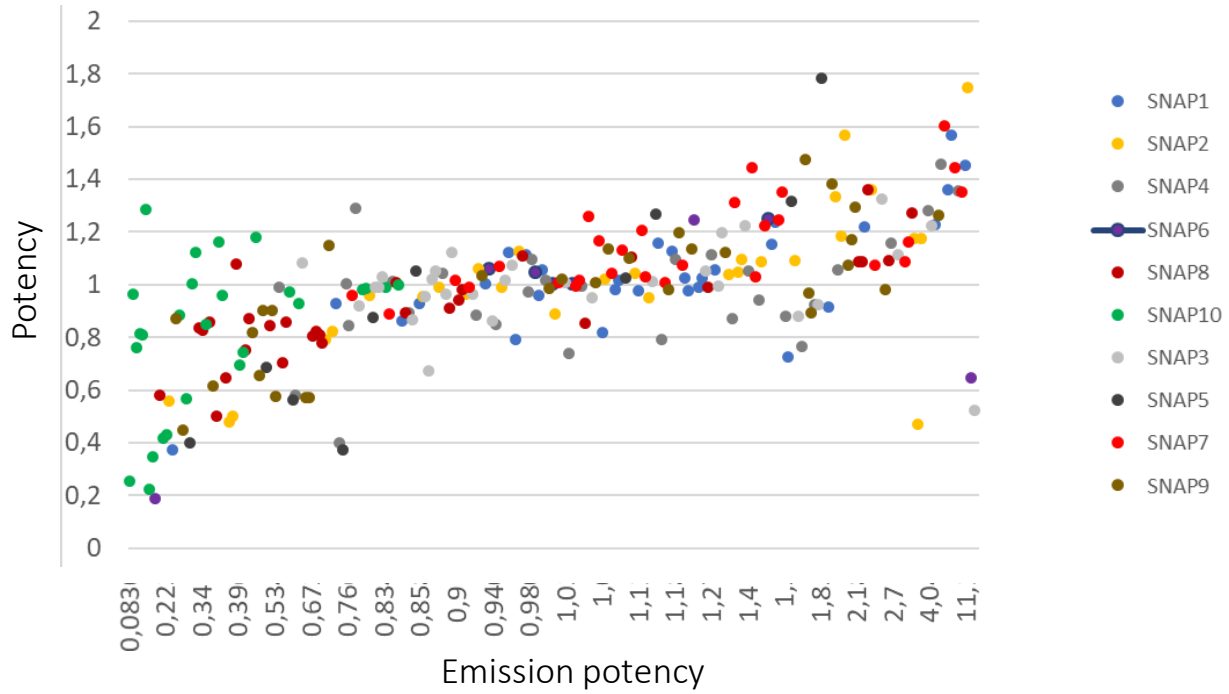


The same figure can be differentiated by pollutant. There is a clear link between potency and emission potency for PPM, whereas potencies are close to one and not clearly dependent on emissions for NO_2. This reflects the fact that NO_2 is a precursor of secondary $PM_{2.5}$ that first needs to be oxidized in the atmosphere and then combined with NH_3 to form particles at a distance to emission sources. Reduction of NO_2 emissions will lead to a wider impact on $PM_{2.5}$ concentrations over the domain. Therefore, the proximity of emission sources to population will not systematically result in an impact close to the population.

A.1.3 Sectoral emission potencies derived with the SHERPA surrogate model

In the Eurodelta II study, potencies were computed for 4 targeted countries by 5 different CTM models to obtain a range of values. The 2014 EEA report used the country specific potencies evaluated by only one of the models (model 3). Specific potencies were used for the 4 countries and an average over these 4 countries was used for the non-evaluated countries (still based only on one model result). Here, the two versions of SHERPA were used to estimate emission potencies: EMEP-SHERPA and CHIMERE-SHERPA. In order to be compared with previous potencies derived in the Eurodelta-II project, emission potencies were calculated for the same 4 countries (France, UK, Spain and Germany). Potencies were also calculated for Italy and Netherlands to extend the analysis. Due to uncertainties in emission inventories, for a given pollutant, potencies are calculated only for sectors that represent more than 2% of the total emissions for this pollutant and if the concentration response due to emission reduction is higher than 2% (sectors for which this is not the case are indicated by "Low. Rep.": "low representativity" in the following tables). Potencies were compared for each country for the industrial sectors SNAP 1, SNAP 3, SNAP 4 and SNAP 9. Only a small number of industrial sites is classified in SNAP 5 and associated emissions are very sporadic. Indeed, for SNAP 5, the errors potentially associated to potency calculation are high, therefore this sector was not taken into account here.

Figure 14: Sherpa based potencies versus emission potency ratio in the emission inventory used in SHERPA



This figure shows that potencies calculated by CHIMERE-SHERPA can be partly explained by the emission potency. For the SNAP 1 sector for which a high potency is calculated (see following paragraph) despite of a low spatial correlation, the figure shows that high potencies are generally explained by high emission potencies. For this sector, even if the majority of emissions occur outside large population centres (low spatial correlation), the high potency derived from CHIMERE-SHERPA (see section A1.3) appears to be influenced by a few hot spots that are only captured when looking at emission potencies.

<u>Potency related to $PM_{2.5}$ exposure:</u>

Table 22 : Sectoral potencies based on CHIMERE-SHERPA, CHIMERE-EMEP and Eurodelta-II results corresponding to $PM_{2.5}$ exposure in Europe due to NO_x emission reductions over selected countries:

NOx potency	**SNAP1 Combustion in energy industries**			**SNAP3 Combustion in manufacturing industry**			**SNAP4 Production processes**			**SNAP9 Waste treatment**		
Country	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta
France	1,03	0,94	0,91	1,01	1,02	0,87	Low Rep.	1,01		Low Rep.	Low Rep.	
Germany	1,06	1,01	0,80	1,05	1,03	0,84	1,00	0,54		Low Rep.	Low Rep.	
Spain	0,82	0,72	0,65	1,12	1,26	0,93	0,99	0,89		Low Rep.	Low Rep.	
United Kingdom	1,03	0,92	0,74	0,96	1,02	0,79	Low Rep.	0,88		Low Rep.	Low Rep.	
Italy	0,93	0,67		0,92	1,25		Low Rep.	Low Rep.		Low Rep.		
Netherlands	0,98	0,97		0,99	1,02		Low Rep.	0,96		Low Rep.		

The NOx potency for $PM_{2.5}$ exposure calculated with SHERPA emission reductions ranges between 0.54 to 1.26 for all industrial SNAPs and the 6 countries investigated. Whereas Eurodelta-II always exhibits

values lower than 1 for SNAP 1 and SNAP 3, values are more dispersed with SHERPA. Generally, EMEP-SHERPA results and CHIMERE-SHERPA results show the same behaviour: higher or lower than one (less true for SNAP3).

Overall, based on SHERPA, the differences between the impacts on exposure of national reductions of NO_x emissions for all sectors and for one single sector specifically do not exceed 30% over the studied countries (except for one case).

Table 23 : Sectoral potencies based on SHERPA results corresponding to $PM_{2.5}$ exposure in Europe due to PPM emission reductions over selected countries.

PPM potency	SNAP1 Combustion in energy industries			SNAP2 Non-industrial combustion			SNAP3 Combustion in manufacturing industry			SNAP4 Production processes			SNAP9 Waste treatment		
Country	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta
France	Low Rep.	1,08	0,64	0,79		1,03	2,04	1,31	0,63	0,89	1,30	1,08	0,45	1,04	
Germany	1,24	1,13	0,51	0,99		1,07	1,22	1,11	0,55	1,11	1,16	1,38	1,12	0,93	
Spain	0,73	0,48	0,39	0,48		1,78	0,88	1,11	0,52	0,76	1,09	0,84	1,38	1,17	
United Kingdom	1,15	0,75	0,47	0,96		1,04	0,86	0,98	0,58	0,97	0,98	1,31	1,20	0,91	
Italy	0,98	0,76		1,13			0,95	1,45		0,87	1,34		0,57	1,12	
Netherlands	1,13			0,82			0,86			0,74			0,98		

For all sectors, the PPM potency for $PM_{2.5}$ exposure calculated with SHERPA, deviates more from unity than for NO_x with a range between 0.45 and 2.04. According to these results, the impact associated to PPM reductions calculated with a homogeneous reduction over all sectors of a given country (as done for the EMEP SRMs) can be underestimated or overestimated by a factor two, depending on sector and country. Eurodelta-II results also show potency further from unity but in Eurodelta-II, potencies lower than one are always found for SNAP1 and SNAP3. One explanation may be the treatment of SNAP 2 emissions. In Eurodelta-II, national emissions from this sector are disaggregated based on population-based proxies. In CHIMERE-SHERPA (and EMEP-SHERPA), an additional parametrisation is used to decrease PPM emissions per inhabitant when the population density increases. This explains SNAP 2 potency values lower than 1 with CHIMERE-SHERPA and higher than 1 for Eurodelta-II. As a mirror effect, this leads to higher potencies for some industrial SNAPs in the CHIMERE-SHERPA case. The same conclusion can be drawn for EMEP-SHERPA (potencies are not shown here).

Here results from the two SHERPA versions are less homogeneous, with several cases where one model shows potencies higher than one and the other lower than one.

Table 24 : Sectoral potencies based on SHERPA results corresponding to $PM_{2.5}$ exposure in Europe due to SO_2 emission reductions over selected countries.

SOx potency	SNAP1 Combustion in energy industries			SNAP3 Combustion in manufacturing industry			SNAP4 Production processes			SNAP9 Waste treatment		
Country	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta
France	0,99	0,98	0,74	1,02	1,04	1,06	0,99	0,83		Low Rep.	Low Rep.	
Germany	1,02	0,98	0,86	1,01	1,00	1,03	1,04	1,08		Low Rep.	Low Rep.	
Spain	0,93		1,01	1,08		1,03	0,88			Low Rep.		
United Kingdom	1,06	0,93	0,86	0,95	1,13	0,96	0,94	1,00		Low Rep.	Low Rep.	
Italy	0,96			0,96			1,10					
Netherlands	1,00			0,99			1,05					

Potencies associated with SO_2 reductions are distributed around unity, ranging between 0.83 and 1.13. This means that, according to SHERPA results, errors associated with reducing emissions over all sectors instead of in a specific one, do not exceed 10%. Potencies around unity have also been calculated with Eurodelta-II.

Table 25: Sectoral potencies based on SHERPA results corresponding to $PM_{2.5}$ exposure in Europe due to NMVOC emission reductions over selected countries.

NMVOC potency	SNAP1 Combustion in energy industries			SNAP3 Combustion in manufacturing industry			SNAP4 Production processes			SNAP9 Waste treatment		
Country	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta	CHIMERE-SHERPA	EMEP-SHERPA	Eurodelta
France	Low Rep.	NA		Low Rep.	Low Rep.		0,85	1,04		Low Rep.	Low Rep.	
Germany	1,16	1,10		Low Rep.	Low Rep.		1,02	1,07		Low Rep.	Low Rep.	
Spain	Low Rep.	NA		Low Rep.	Low Rep.		0,58	0,96		1,14	0,97	
United Kingdom	Low Rep.	0,72		Low Rep.	Low Rep.		0,40	0,87		0,66	0,92	
Italy	Low Rep.						0,85					
Netherlands	0,98						0,79					

For several countries, potencies associated with NMVOC are not calculated for SNAP 1 and SNAP 3 because these sectors represent less than 2% of total NMVOC emitted. Overall potencies lower than one are calculated with CHIMERE-SHERPA and EMEP-SHERPA meaning that reducing NMVOC over industrial sectors is less efficient than a homogeneous reduction targeting all sectors. This has probably to do with the fact that the main sector emitting NMVOCs is the residential sector (household solvents, SNAP6) for which emissions are directly spatialized based on a population proxy, and so are very well correlated with population.

Overall, the largest differences between the impact of emission reductions in specific sectors and homogeneous reductions over all sectors are logically found for primary particulate matter (PPM). PPM emission reductions mainly impact the local vicinity of industry, whereas secondary pollutants form over larger distances and have a broader impact reducing the differences in efficiency between "all-sectors" and reductions in individual sectors.

Differences observed between the two SHERPA versions and Eurodelta-II can be partly explained by differences in the emission spatialization and the distribution of emissions over all sectors.

Potency related to NO_2 exposure:

Potencies were also calculated with CHIMERE-SHERPA for NO_x emissions and NO_2 exposure, even though the spatial representativeness of CHIMERE-SHERPA remains only relevant for the calculation of background NO_2 given the spatial resolution of 7km. The impact of reducing emissions of all the other pollutants is not significant for NO_2 exposure. For information, potencies calculated for the road sector (SNAP 7) are also shown.

Table 26 : Sectoral potencies based on CHIMERE-SHERPA results corresponding to NO_2 exposure in Europe due to NO_x emission reductions over selected countries.

Country	SNAP1	SNAP3	SNAP4	SNAP7
France	0,90	0,91	Low. Rep.	1,08
Italy	0,69	0,66	Low. Rep.	1,11
United Kingdom	0,90	0,97	Low. Rep.	1,05
Netherlands	0,84	0,80	Low. Rep.	1,05
Spain	0,47	0,77	0,60	1,34
Germany	1,00	1,01	0,84	1,01

Potency values for industrial sectors are mainly lower than 1, showing values between 0,47 and 1,01, whereas potencies calculated for the road sector (SNAP 7) are higher than one for all countries. This is explained by the fact that NO_2 emissions are dominated by the road sector, which is more correlated with population density than industrial activities. Therefore, a homogeneous reduction over all sectors will mainly lead to a reduction of road transport emissions and will have a larger impact on exposure than reductions over a specific industrial sector.

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