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Costs of air pollution from European industrial facilities 2008–2012

— an updated assessment

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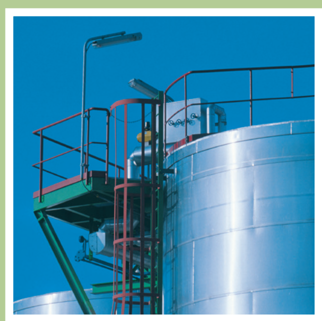

European Environment Agency

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Executive summary

In 2011, the European Environment Agency (EEA) published a first assessment of the costs of air pollution caused by European industrial facilities. The report *Revealing the costs of air pollution from industrial facilities in Europe* (EEA, 2011) applied a simplified modelling approach to assess the damage costs to health and the environment in 2009, caused by pollutant emissions from industrial facilities officially reported to the European Pollutant Release and Transfer Register (E-PRTR) ([1]).

Since 2011, the annual assessments of Europe's air quality published by EEA have regularly concluded that, despite a number of past successes in reducing emissions, air quality still needs to improve in order to reduce harm to human health and the environment ([2]). The need for regularly updated knowledge concerning air pollution sources, the subsequent levels of human and environmental exposure, and its associated costs remains important.

This report presents an updated assessment of the cost of damage to health and the environment in monetary terms from air pollution released in the years 2008 to 2012 by industrial facilities in the EU-27, Norway and Switzerland. The approach employed to estimate damage costs is again based upon existing standard policy tools and methods, such as those originally developed under the EU's Clean Air for Europe (CAFE) programme for the main air pollutants and since updated during the recent review of the European Union's (EU) air pollution policies performed by the European Commission. The assessment also uses other existing models and approaches used to inform policymakers about the damage costs for other pollutants. Together, the methods are used to quantify the impacts and associated damage costs caused by a number of pollutants emitted from industrial facilities, including:

- the main air pollutants: ammonia (NH_3), nitrogen oxides (NO_X), non-methane volatile organic compounds (NMVOCs), particulate matter (PM_{10}) and sulphur oxides (SO_X);
- heavy metals: arsenic, cadmium, chromium, lead, mercury and nickel;
- organic compounds: benzene, dioxins and furans, and polycyclic aromatic hydrocarbons (PAHs);
- carbon dioxide (CO_2).

Each of these pollutants can harm human health, the environment or both. Certain of them contribute to forming ozone and particulate matter in the atmosphere. There are significant differences in terms of the extent of current knowledge between the selected pollutants and the methods available to estimate their respective impacts.

Key findings

The aggregated cost of damage over the period 2008–2012 caused by emissions from the E-PRTR industrial facilities is estimated as being at least EUR_{2005} 329 billion (and up to EUR_{2005} 1 053 billion) ([3]). Table ES.1 shows the damage costs in each year for the different pollutants assessed in this report.

Across the five-year period as a whole, information was available for a total of 14 325 individual facilities ([4]). Damage costs from these facilities decreased during the period. Various factors will have contributed to this decrease, including the ongoing impacts of environmental legislation and the economic recession in Europe which resulted in lower rates of industrial activity in years immediately after 2008. The majority of the quantified damage costs is caused by emissions of the main air pollutants and CO_2. While damage cost estimates associated with heavy metal and organic pollutant emissions are significantly lower, they still contribute hundreds of millions of euros harm to health and the environment, and at the local scale can cause significant adverse impacts.

([1]) http://prtr.ec.europa.eu.
([2]) e.g. *Air quality in Europe — 2013 report* (EEA, 2013a).
([3]) Damage cost estimates provided throughout the report are expressed in 2005 euros.
([4]) In instances where a facility changes ownership, the E-PRTR register records this as a new facility in the register. Over the 5-year period, a certain number of E-PRTR facilities will also have ceased operating while other new facilities will have commenced operations.

Table ES.1 Estimated damage costs aggregated by pollutant group, 2008–2012 (2005 prices)

Pollutant group	Aggregated damage cost (billion EUR_{2005})				
	2008	2009	2010	2011	2012
Main air pollutants (NH_3, NO_X, PM_{10}, SO_2, NMVOCs)	58–168	47–136	44–129	43–124	40–115
CO_2	20–82	18–73	19–76	18–74	18–73
Heavy metals (As, Cd, Cr, Hg, Ni, Pb)	0.53	0.34	0.43	0.34	0.34
Organic pollutants (benzene, dioxins and furans, PAHs)	0.22	0.11	0.17	0.22	0.10
Sum	**79–251**	**65–209**	**64–206**	**62–199**	**59–189**

For the main air pollutants and CO_2, damage costs are expressed as a range. This reflects that for these pollutants, different methods or assumptions are used in the calculations. Furthermore, expressing damage costs as a range helps illustrate the often considerable uncertainty which is inherent in such analyses.

- For the main air pollutants, the range provided corresponds to the use of two contrasting but complementary approaches for valuing health damage — the value of a life year (VOLY), and a (higher) value of statistical life (VSL) (e.g. OECD, 2012). This report's analysis for the main air pollutants extends to quantifying crop and building material damage from these pollutants but does not include their negative impacts on ecosystem services, such as harm to biodiversity, which in some instances may be significant ([5]). This implies that the damage costs are therefore likely to be under-estimated.

- The ranges shown for CO_2-related damage costs reflects the difference between the minimum (EUR_{2005} 9.5 per tonne CO_2) and maximum values (EUR_{2005} 38.1 per tonne CO_2) used in this report for carbon valuation. The present report applies a similar but more nuanced approach to valuing CO_2-related damage costs than in the previous 2011 report, in which only a single value for CO_2-related damage costs was used (EUR 33.6) based upon a method applied at that time by the UK government for valuing carbon emissions. The selected values used in the present report are based upon carbon price values for the EU Emissions Trading System (ETS) used in policy modelling by the European Commission. This approach provides a reflection of the costs associated with decreasing CO_2 emissions over time in line with the required reduction necessary to meet the current policy objective of limiting future limit average global surface temperature increase to two-degrees.

Care is needed when interpreting the results. The E-PRTR Regulation (EU, 2006) requires only those industrial facilities with an activity rate exceeding a defined threshold and emissions exceeding the pollutant-specific thresholds to report information to the register. As a result, the E-PRTR's coverage varies significantly across the different pollutants and sectors. The total cost of damage to health and the environment from all sectors of the economy, including from 'diffuse' sources such as road transport and households, and from all pollutants will therefore be significantly higher than the estimates presented here. The European Commission has, for example, recently estimated that in 2010, the external costs associated with only the main air pollutants were in the range of EUR 330–940 billion (European Commission, 2013a).

As observed in the 2011 report, a limited number of industrial facilities cause the vast majority of the damage costs to health and the environment. Fifty per cent of the total damage cost occurs as a result of emissions from just 147 (or 1 %) of the 14 325 facilities that reported data for releases to air during this period (Figure ES.1 and Map ES.1). Three quarters of the total damage costs were caused

([5]) *Impacts of air pollution on ecosystems, human health and materials under different Gothenburg Protocol scenarios* (WGE, 2012).

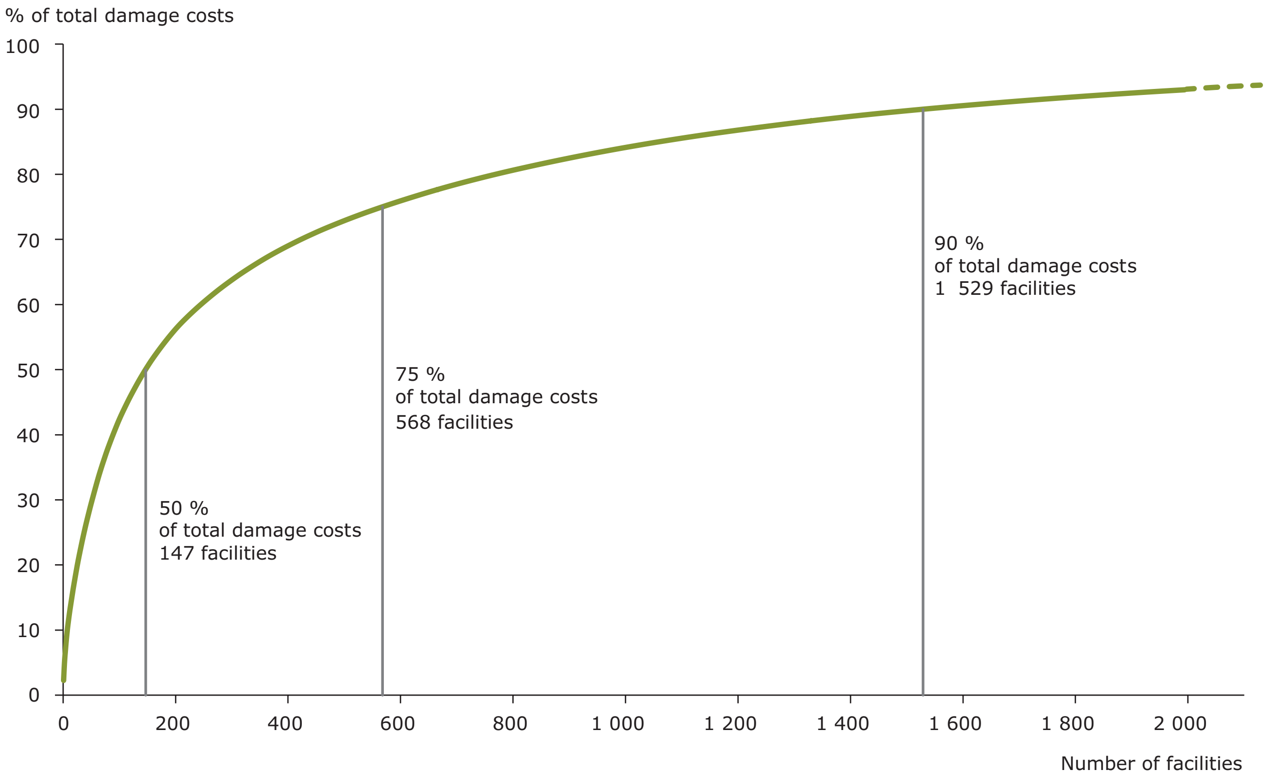
by the emissions of 568 facilities (4 % of the total number of facilities), and 90 % of damage costs are attributed to 1 529 facilities (11 % of the total). These findings should, however, not take focus away from the need to also regulate emissions from smaller facilities, which on the local scale can contribute significantly to air pollution and its subsequent harmful impacts.

The report lists the top 30 individual facilities identified as causing the highest damage across the five-year period 2008–2012. Of these, 26 are power-generating facilities, mainly fuelled by coal/lignite and located predominantly in Germany and Eastern Europe. Not surprisingly, most of the facilities with high emission damage costs are among the largest facilities in Europe, releasing the greatest amount of pollutants.

A simple ranking of facilities according to their aggregate emission damage costs provides little indication, however, of the relative efficiencies of production. To illustrate this, the differences in environmental efficiencies of power generating facilities were assessed using the reported CO_2 emissions from as a proxy for fuel consumption. One difference noted when damage costs from these facilities are normalised by CO_2 emissions is that more power generating facilities from eastern Europe appear at the top of the results, indicating they are less environmentally efficient and relatively more damaging to health and the environment.

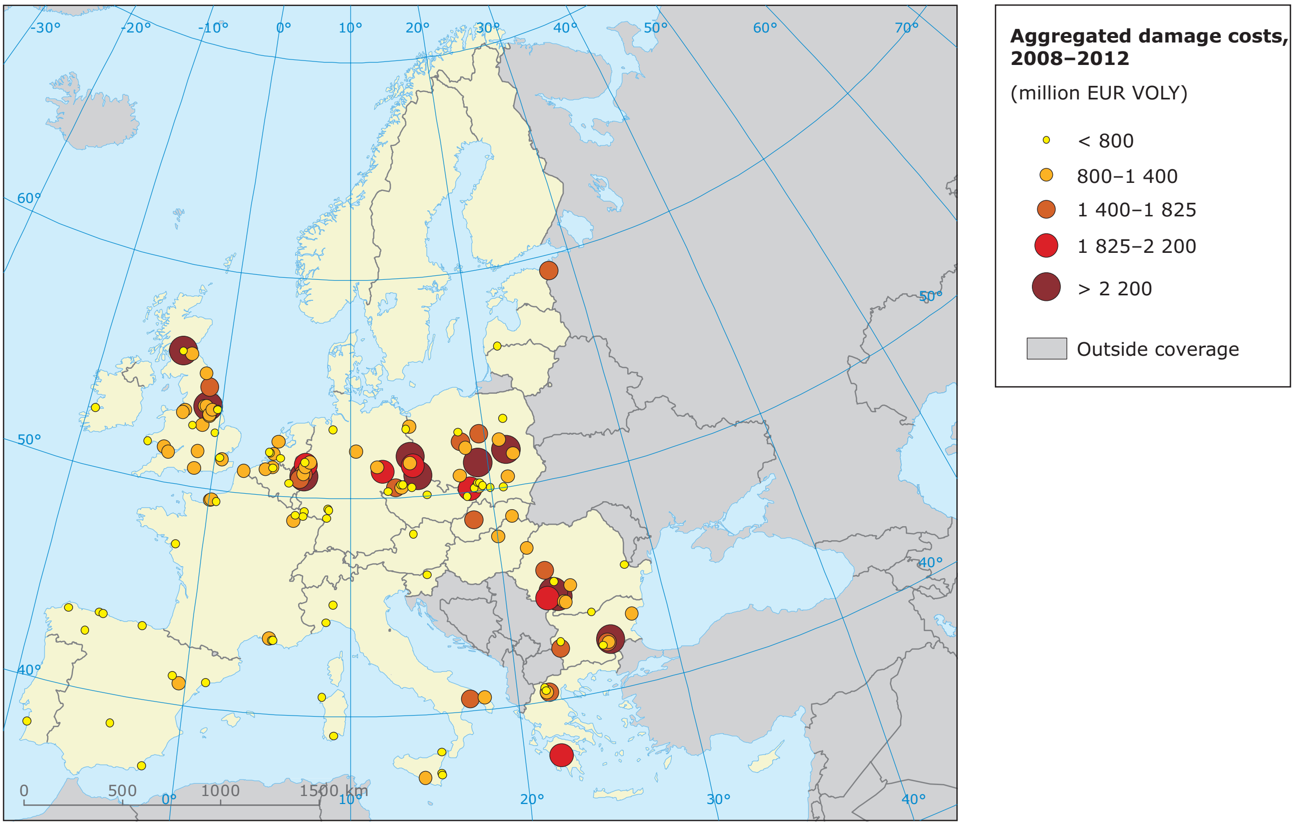
Of all the industrial sectors included in the E-PRTR pollutant register, emissions from the energy sector contributed the largest share of the damage costs across the five-year period assessed

Figure ES.1 Cumulative distribution of the estimated damage costs associated with emissions of selected pollutants from E-PRTR facilities, 2008–2012



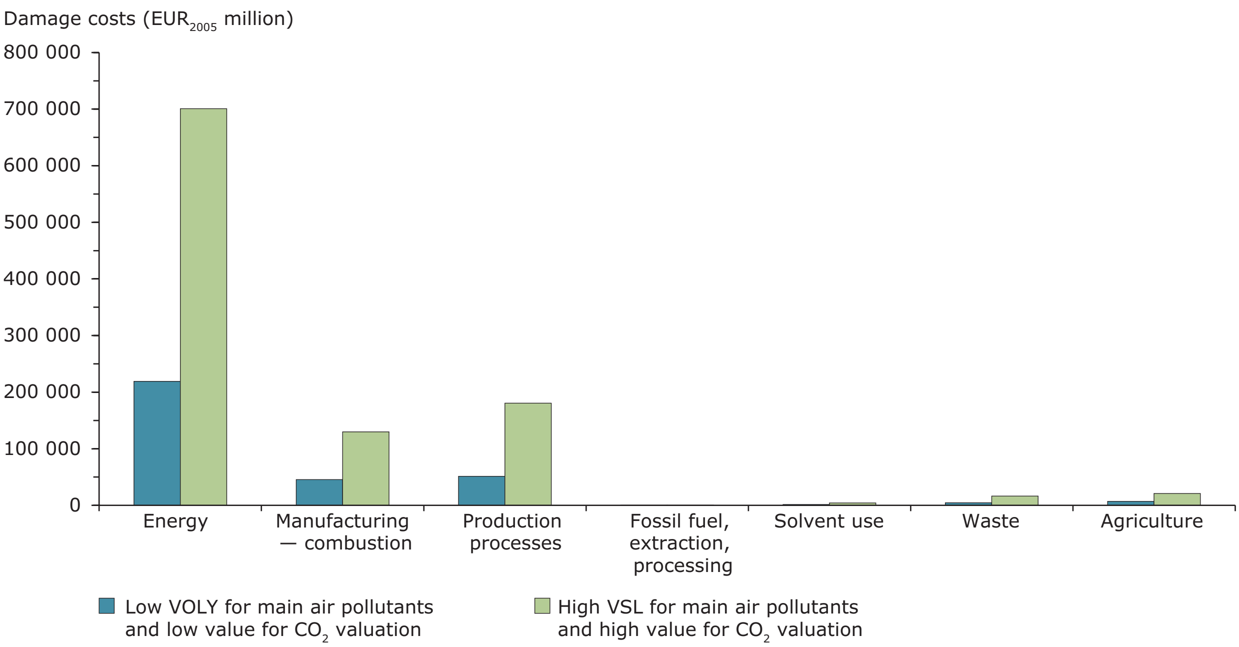
Note: The distribution is based on the lower VOLY approach for the main air pollutants and a CO_2 price of EUR_{2005} 9.5 per tonne.

Map ES.1 Location of the 147 E-PRTR facilities that contributed 50 % of the total damage costs estimated for 2008–2012



Note: The lower VOLY approach for the main air pollutants and a CO_2 price of EUR_{2005} 9.5 per tonne are applied.

Figure ES.2 Aggregated damage costs by sector, 2008–2012



Note: The low-high range shows the differing results derived from the alternative approaches to (a) mortality valuation for the main air pollutants and (b) the difference between minimum (EUR_{2005} 9.5 per tonne CO_2) and maximum values (EUR_{2005} 38.1 per tonne CO_2) used in this report for carbon valuation.

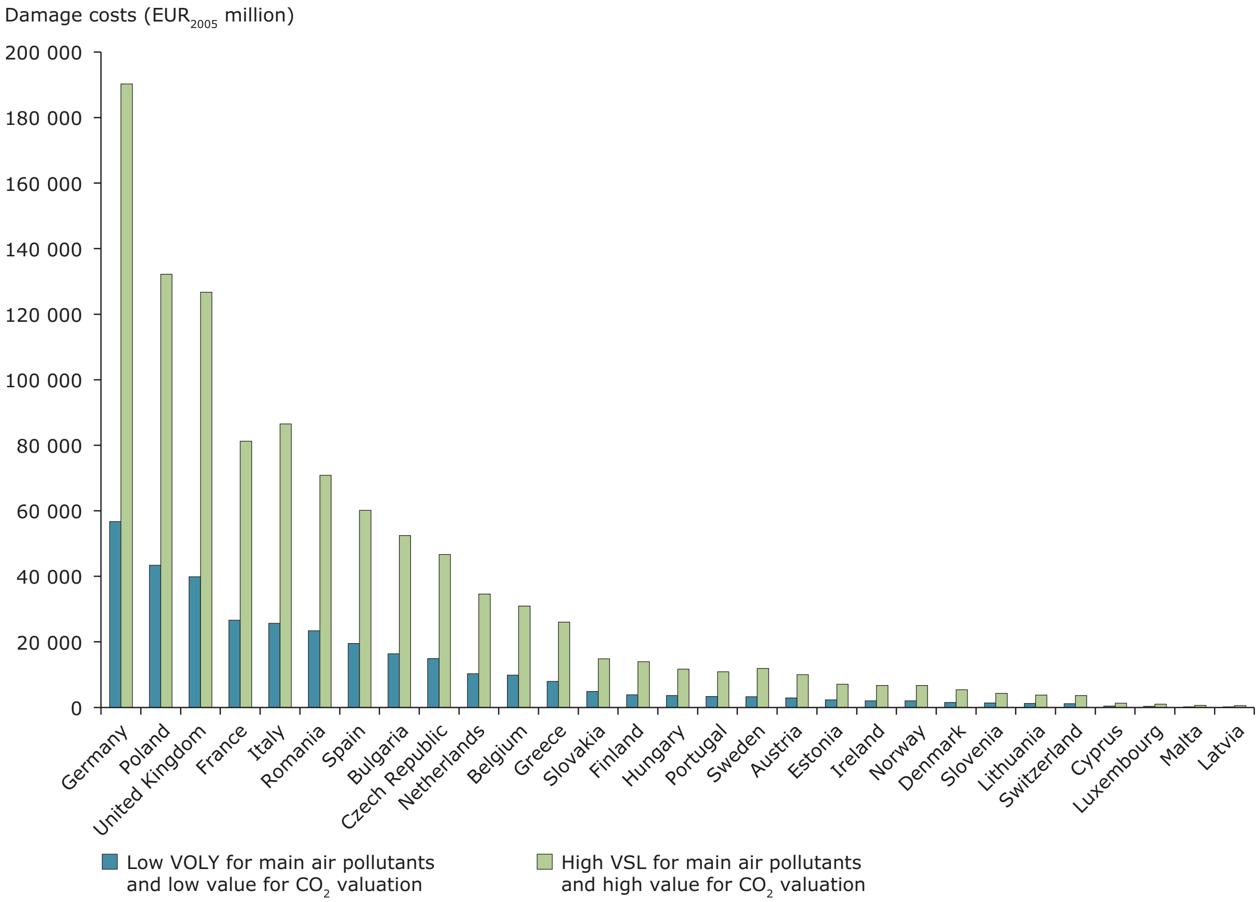
(estimated as at least EUR_{2005} 219 billion (and up to EUR_{2005} 701 billion) (Figure ES.2). Sectors involving production processes and combustion used in manufacturing were responsible for most of the remaining estimated damage costs.

Results aggregated by country are shown in Figure ES.3. Not surprisingly, countries such as Germany, Poland, the United Kingdom, France and Italy, which have a high number of large facilities, contribute the most to total estimated damage costs.

As an alternative to weighting damage costs by CO_2 emissions as was done for individual facilities, GDP can be used as an indicator of national production to normalise the national damage costs against the respective level of services provided/generated by the national economies (Figure ES.4). When applying this measure, certain countries previously shown as having the highest damage costs — Germany, the United Kingdom, France and Italy — drop significantly down the ranking, while Bulgaria, Romania and Estonia, rise to the top. Poland remains toward the top of the rankings, reflecting the high amounts of pollutants at Polish facilities emitted relative to national gross domestic product.

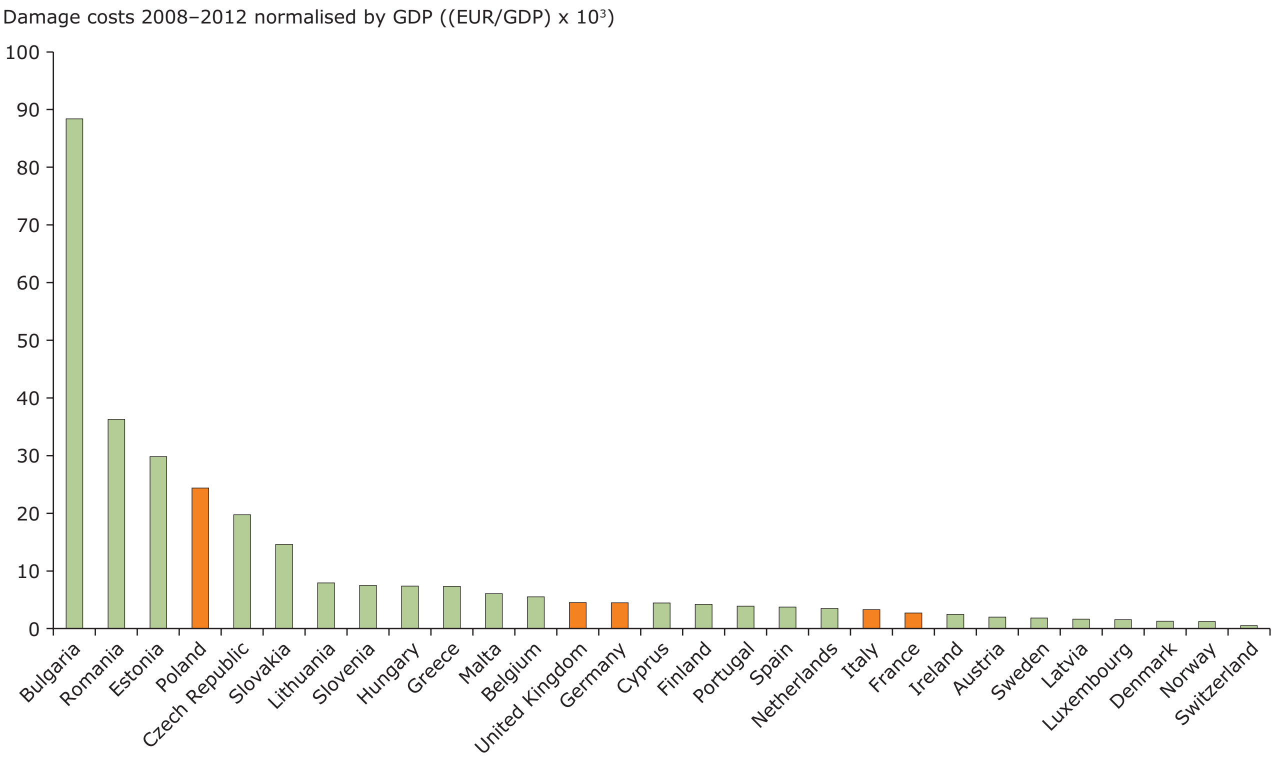
Finally, as an example of the wider application of the methods developed for estimating air pollution related damage costs, this report shows that if 1 500 large combustion plants in Europe were hypothetically all to meet the emission limit values set in the Industrial Emissions Directive (EU, 2010) for just NO_X and SO_2, the direct benefits

Figure ES.3 Aggregated damage costs by country, 2008–2012



Note: The low-high range shows the differing results derived from the alternative approaches to a) mortality valuation for the main air pollutants and b) the difference between minimum (EUR_{2005} 9.5 per tonne CO_2) and maximum values (EUR_{2005} 38.1 per tonne CO_2) used in this report for carbon valuation.

Figure ES.4 Aggregated damage costs by country normalised against GDP, 2008–2012



Note: The orange bars highlight the countries with the highest absolute damage costs in Figure ES.2. The ranking is based on the lower VOLY approach for mortality valuation for the main air pollutants and a CO_2 price of EUR_{2005} 9.5 per tonne.

in the EU-27 would be in the order of at least EUR_{2005} 11.2 billion (and up to EUR_{2005} 32.7 billion) per year. This is calculated by coupling the damage costs methodology with the hypothetical emission reduction documented in another recent EEA assessment (EEA, 2013b). In reality, those savings would be significantly greater, as savings would also occur as a result of reduced emissions of other pollutants that were not quantified (e.g. PM_{10}, NMVOCs, heavy metals and organic pollutants). It is clear that regardless of the choice of damage cost values and methodologies employed, substantial health and environmental benefits would result if emissions of pollutants were to reduce from industrial facilities in the future.

New elements of the updated assessment

There are a number of new or updated aspects addressed in the present report compared to the earlier 2011 assessment. These include:

- *Application of new science:* the most significant methodological change compared to the 2011 report is an updated methodology to take into account recent air quality model and damage cost developments for the main air pollutants.

- *Data for five consecutive years:* The updated methodology has been applied to E-PRTR data from five years — 2008 to 2012.

- *Carbon pricing approaches:* There remains a wide range of approaches used to estimate damage costs associated with CO_2 or to quantify the benefit of CO_2 emission reductions for policy assessment purposes. There is, as yet, no established methodology for this, unlike the situation for the main air pollutants. Uncertainties and limitations associated with these approaches are high. Examples of such approaches include those based upon modelled carbon price forecasts, the social cost of carbon (SCC), marginal abatement costs etc. The present report applies a range of values based upon recent modelled EU Emissions Trading System (ETS) carbon price forecasts performed for the European Commission to support the proposal for a 2030 climate and energy policy framework.

- *Potential damage costs savings:* The present report draws on the results of a recent EEA assessment, which investigated the hypothetical emission reduction potential of NO_X, SO_2 and dust from more than 1 500 of Europe's large combustion plants, to illustrate the scale of associated savings in terms of reduced damage costs.

As with the 2011 EEA report, the present report does not assess whether the emissions of a given facility are consistent with its legal permitted conditions for operating. Furthermore, while presenting the damage costs for human health and the environment from industrial facilities, the report again does not assess the recognised benefits of industrial facilities (such as the production of goods and products, and generating employment and tax revenues). It is important that such benefits of industrial activity are properly recognised, but such an assessment is beyond the scope of this report.

Recommendations

The report identifies several important ways in which the E-PRTR and its implementation might be improved for use in assessment studies. These include:

- **Better completeness of emissions from individual facilities**. A number of instances were identified in the course of this updated assessment where it seems clear that certain facilities are not reporting emissions of certain pollutants which are expected to occur above the release thresholds set in the E-PRTR Regulation. Member States should further improve the quality checking of facility information before it is reported to the E-PRTR, particularly to address completeness of data and identify outlying values.

- **Providing information on the fuel consumption or productive output of individual facilities**. This would enable the environmental efficiency of facilities to be calculated in terms of estimated damage costs per unit of production or fuel consumption, and allow an increased focus upon resource efficiency. It would also facilitate independent verification of data reported to the register. While the Large Combustion Plant Directive (2001/80/EC) requires Member States to report information on fuel used in the plants, linking these data to the E-PRTR information is difficult.

- **Improved traceability of facilities**. Comparing the present study's results with those of previous studies on a facility-by-facility basis was difficult. While some older facilities may have closed since these earlier studies were performed, part of the problem relates to differences in the annual E-PRTR datasets received by the EEA. Facilities often change ownership, name, and/or national facility identification code, creating difficulties in linking the annually reported emissions. Similarly, linking E-PRTR data with information reported under other EU legislation such as the Large Combustion Plant Directive is difficult, due to differences in facility definitions, facility names and identifiers etc. Improved streamlining of information reported under EU legislation would very much benefit assessment activities, while also providing additional means for the verification of official data and potentially reducing the reporting requirements for countries. It is noted in this context that the European Commission is presently undertaking work with a view to ensure the future linking of information reported on large combustion plants with E-PRTR, as well as for streamlining of reporting between the Industrial Emissions Directive (IED) and E-PRTR.

In summary, this report presents an updated methodology that allows for the estimation of damage costs caused by emissions of selected pollutants from industrial facilities included in the E-PRTR. It demonstrates that, compared to using emissions data alone, these methods provide additional insights and transparency into the costs of harm caused by air pollution. The sensitivity of the results to the choice of pollutant specific damage cost values used, as well as the importance of normalising results to take into account a measure of the efficiency of production across the different industrial facilities, is demonstrated. Such insights are particularly valuable in the context of current discussions in Europe on how best to move towards a resource-efficient and low-carbon economy.

1 Introduction

1.1 Background

In 2011, the European Environment Agency (EEA) published a technical report *Revealing the costs of air pollution from industrial facilities in Europe* (EEA, 2011). The report assessed the damage costs to health and the environment in 2009, caused by pollutant emissions from industrial facilities officially reported to the European Pollutant Release and Transfer Register (E-PRTR — Box 1.1).

Knowledge of the magnitude of emissions released from a specific industrial facility, as is available in E-PRTR, does not in itself provide information on the subsequent impacts of these pollutants on human health and the environment, nor the associated monetary costs of such damage. An application of modelling frameworks that link knowledge of pollutant emissions with their impacts and consequent damage costs is therefore necessary.

There has been significant research undertaken to develop improved scientific modelling frameworks and economic methods for estimating the impacts and damage costs of air pollution. Such methods have been developed through research funded by the European Commission and Member States since the early 1990s (e.g. Holland et al., 2005a and

Box 1.1 The European Pollutant Release and Transfer Register (E-PRTR)

The E-PRTR — established by the E-PRTR Regulation (EU, 2006) — provides information on releases of 91 different pollutants to air, water and land from around 28 000 industrial facilities in 27 Member States of the European Union (Croatia has not yet reported data to E-PRTR since its accession to the European Union in July 2013), Iceland, Liechtenstein, Norway, and from 2010, Serbia and Switzerland (E-PRTR, 2013). Each year, around 9 800 facilities report at least some information on releases to air. Not all facilities are required to report data each year (see below). Around 9 800 facilities report emissions of pollutants to air annually; the remaining facilities report pollutant releases to water, soil and/or transfers of waste. For the EU, the Register implements the UNECE (United Nations Economic Commission for Europe) PRTR Protocol to the Aarhus Convention on Access to Information, Public Participation in Decision-making and Access to Justice in Environmental Matters.

The E-PRTR provides environmental regulators, researchers and the public with information about pollution released from industrial farms, factories and power plants. It demonstrates that national regulators are aware of the size of emissions from specific facilities within their jurisdictions. By focusing on releases to the environment, the E-PRTR is instrumental in addressing potential burdens on health and the environment in a way that can be quantified using well-established methods. A further strength is that data are updated annually. This allows year on year comparisons of the emissions from individual facilities to be made by the public, with information of whether emissions are rising or falling.

One of the main weaknesses of the register, however, is the absence of any requirement to report activity data in the submitted information i.e. the specific fuel combustion or productive output from facilities. This severely restricts the use of the register for further analytical or assessment purposes, and weakens the possibility for EU authorities to perform meaningful quality checks and/or verification of the reported emissions. Further, the differences in technical definitions between E-PRTR facilities and other industrial reporting undertaken under separate European Union legislation (e.g. Large Combustion Plant Directive (2001/80/EC), Industrial Emissions Directive (2010/75/EU), and the European greenhouse gas Emissions Trading System (2003/87/EC and subsequent amendments)) greatly complicate efforts to supplement E-PRTR data with information reported elsewhere.

The European Commission has recently reported on the first three years of implementation of the E-PRTR to the European Parliament and the Council (European Commission, 2013b). As part of the review, several options to improve the E-PRTR were identified, including actions to enhance the quality and completeness of the pollutant release data across all environmental media.

2005b; Hurley et al., 2005) and have been subject to international peer review (e.g. Krupnick et al., 2005). Methods such as those developed under the European Commission's Clean Air for Europe programme (CAFE) are regularly applied in cost-benefit analyses to support national, EU and international policymaking in air pollution and climate mitigation.

In addition to the CAFE programme, these methods for determining environmental externalities associated with air pollution have also been applied to inform the development of a considerable amount of European environmental policy and international agreements, including:

- The National Emission Ceilings Directive (EU, 2001b), setting total emission limits for sulphur dioxide (SO_2), nitrogen oxides (NO_X), ammonia (NH_3) and non-methane volatile organic compounds (NMVOCs) for EU Member States, and the related Gothenburg Protocol to the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP Convention) (UNECE, 1999) and later amendment; (e.g. Pye et al., 2007, Holland et al., 2011);
- The Air Quality Directives (EU, 2004a and 2008), setting concentration limits for pollutants in the ambient air (e.g. AEA Technology, 1997; Holland and King, 1998, Entec, 2001; Holland et al., 2001; Holland et al., 2005c);
- The proposal for a new Clean Air Policy Package for Europe (European Commission, 2013a, 2013c; Holland, 2014b);
- EU climate change mitigation policies where the co-benefits of reducing air pollutants are typically quantified e.g. the Roadmap for moving to a competitive low carbon economy in 2050 (European Commission, 2011a);
- The Large Combustion Plant Directive (EU, 2001a), feeding into the Industrial Emissions Directive (EU, 2010);
- The revision of the Fuel Quality Directive (EU, 1999 and 2003; e.g. Bosch et al., 2009);

There are acknowledged uncertainties in the scientific knowledge and modelling framework that underpins the assessment of damage costs arising from air pollution. For example, such methods cannot yet provide quantification for all types of damage, particularly those relating to ecosystems such as the harm caused to biodiversity. Methods are also still evolving as the scientific knowledge base improves, so calculated estimates of damage costs are not considered to be as 'accurate' as the emissions data. However, despite these uncertainties, it is possible to quantify a number of impacts and subsequent damage costs for a range of pollutants.

The 2011 EEA report coupled reported emission data with existing standard policy tools and methods to determine the related environmental externalities. In addition to applying methods based on the CAFE approach to assess damage costs arising from the emission of the 'traditional' main air pollutants (e.g. NO_X, SO_2, particulate matter (PM), etc., see Box 1.3), the report also estimated the damage costs caused by emissions of heavy metals, organic pollutants and the greenhouse gas carbon dioxide (CO_2). This was done again through the use of existing models and approaches that were in use at the time to inform European and national policymakers about the damage costs of these pollutants.

Such quantification, in monetary terms, of the cost of damage to health and the environment from air pollution subsequently enabled a variety

Box 1.2 General principles in assessing environmental externalities

In order to account for the external costs of air pollution, an individual pollutant's adverse impacts on human health and the environment are expressed in a common metric (a monetary value). Monetary values have been developed through cooperation between different scientific and economic disciplines, linking existing knowledge in a way that allows external costs to be monetised.

Damage costs incorporate a certain degree of uncertainty. However, when considered alongside other sources of information, damage costs can support decisions by drawing attention to the implicit trade-offs inherent in decision-making e.g. in cost benefit analysis used to inform legislative impact assessments.

of questions to be addressed in the 2011 report, including the following.

- Which industrial sectors and countries included in E-PRTR contributed most to the estimated damage costs of air pollution in Europe?
- How many facilities accounted for the largest share of air pollution's estimated damage costs?
- Which individual facilities reporting to the E-PRTR pollutant register were responsible for the highest estimated damage costs?

1.2 Scope of this report — what is new

This report updates the earlier assessment of the costs of air pollution from European industrial facilities presented in EEA (2011). There remains no single method available to estimate damage costs for all the pollutant groups addressed in the report (main air pollutants, selected heavy metals and organic pollutants, and CO_2). Aggregating results derived from the different approaches therefore continues to pose challenges and contributes to the uncertainties inherent in an assessment of environmental damage costs. However, one of the key advantages of assessing damage costs using a common measure (i.e. monetary value) is that it exactly enables different types of damage to be aggregated, providing an insight into the total damage costs to health and the environment caused by releases of pollutants into the atmosphere.

Updated aspects of this report include:

- *Application of new science:* The most significant methodological change compared to the 2011 report is an updated methodology to take into account recent air quality model and damage cost developments for the main air pollutants. These changes directly affect the damage costs calculated for individual facilities.
- *Data for five consecutive years:* The updated methodology has been applied to E-PRTR data from five years — 2008 to 2012.
- *Carbon pricing approaches:* The 2011 report used a single value to estimate CO_2-related damage costs, based upon a target-consistent approach used at that time by the UK government for policy impact assessments. The present report now applies a range for CO_2-related damage costs based upon modelled carbon price forecasts for the EU Emissions Trading System (ETS).
- *Accounting for efficiency*: The 2011 report highlighted the need that different operating efficiencies be considered when damage costs from individual facilities are compared. It is clear that certain facilities have high damage cost estimates simply because of their size and production or activity levels e.g. large power stations. It is possible that such large facilities may be more efficient and cleaner than a number of smaller facilities that together deliver the same level of service or output. The opposite may also be true. Given the lack of facility fuel-use information in the data reported under E-PRTR, which would allow a more rigorous normalisation to be performed, this report explores several proxy methods of normalising damage costs in an attempt to take into account the efficiency differences between facilities.
- *Potential damage cost savings:* A recent assessment published by the EEA investigated the hypothetical emission reduction potential of NO_X, SO_2 and dust from more than 1 500 of Europe's large combustion plants that operated in 2009 (EEA, 2013b). The present report draws on the results from that study to illustrate the scale of potential benefits, in terms of reduced damage costs, which would occur if certain facilities were to reduce future levels of emissions.

Finally, as with the 2011 EEA report, the present report does not assess whether the emissions of a given facility are consistent with its legal permitted conditions for operating when using E-PRTR data and calculating damage costs from individual facilities. Furthermore, the report does not assess the recognised benefits of industrial facilities (such as the production of goods and products, and generating employment and tax revenues) when presenting the damage costs for human health and the environment from industrial facilities,

Box 1.3 Air pollutants included in this study and their effects on human health and the environment

Nitrogen oxides (NO_x)
Nitrogen oxides are emitted from fuel combustion, such as from power plants and other industrial facilities. NO_x contributes to acidification and eutrophication of waters and soils, and can lead to the formation of particulate matter and ground-level ozone. Of the chemical species that comprise NO_x, it is NO_2 that causes adverse effects on health; high concentrations can cause airway inflammation and reduced lung function.

Sulphur oxides/sulphur dioxide (SO_x/SO_2)
Sulphur dioxide is emitted when fuels containing sulphur are burned. As with NO_x, SO_2 contributes to acidification, with potentially significant impacts including adverse effects on aquatic ecosystems in rivers and lakes, and damage to forests. High concentrations of SO_2 can affect airway function and inflame the respiratory tract. SO_2 also contributes to the formation of particulate matter in the atmosphere.

Ammonia (NH_3)
Ammonia, as for NO_x, contributes to both eutrophication and acidification. The vast majority of NH_3 emissions — around 93 % in Europe — come from the agricultural sector. A relatively small amount is also released from various industrial processes, transportation and waste management.

Non-methane volatile organic compounds (NMVOCs)
NMVOCs, important ground-level ozone precursors, are emitted from a large number of sources including industry, paint application, road transport, dry-cleaning and other solvent uses. Certain NMVOC species, such as benzene (C_6H_6) and 1,3-butadiene, are directly hazardous to human health.

Particulate matter (PM)
In terms of potential to harm human health, PM is one of the most important pollutants as it penetrates into sensitive regions of the respiratory system, and can cause or aggravate cardiovascular and lung diseases and cancers. PM is emitted from many sources and is a complex mixture comprising of both primary and secondary PM; primary PM is the fraction of PM that is emitted directly into the atmosphere, whereas secondary PM forms in the atmosphere following the release of precursor gases (mainly SO_2, NO_x, NH_3 and some NMVOCs).

Heavy metals
The heavy metals arsenic (As), cadmium (Cd), chromium (Cr) lead (Pb), mercury (Hg) and nickel (Ni) are emitted mainly as a result of various combustion processes and from industrial activities. As well as polluting the air, heavy metals can be deposited on terrestrial or water surfaces and subsequently build up in soils and sediments. Heavy metals can also bio-accumulate in food chains. They are typically toxic to both terrestrial and aquatic ecosystems.

Organic pollutants
Benzene, polycyclic aromatic hydrocarbons (PAHs), and dioxins and furans are categorised as organic pollutants. They cause different harmful effects to human health and ecosystems, and each of these pollutants is a known or suspected human carcinogen. Dioxins and furans and PAHs also bio accumulate in the environment. Emissions of these substances commonly occur from the combustion of fuels and wastes and from various industrial processes.

Carbon dioxide (CO_2)
Carbon dioxide is emitted as a result of the combustion of fuels such as coal, oil, natural gas and biomass for industrial, domestic and transport purposes. CO_2 is the most significant greenhouse gas influencing climate change, thereby posing a threat to public health and the environment.

2 Methods

An overview of the methods used and further detail on the approaches employed to quantify the benefits of reducing emissions of main air pollutants, heavy metals and organic compounds, and greenhouse gases are presented in this chapter.

In the past there has been extensive debate about the methods used to estimate impacts and associated damage costs of main air pollutants under the CAFE Programme, and some consensus has been reached in this area. There has been less debate, however, about the approaches used for the estimating the impact and damage costs from heavy metals, organic pollutants and CO_2. As a result, the methodology for these pollutants may be considered less robust. The methodological description in this section largely draws upon that provided in EEA (2011), updated where relevant.

2.1 The impact pathway approach

The analyses presented here for all pollutants, except CO_2, are based on the Impact Pathway Approach (IPA). This was originally developed in the 1990s in a collaborative programme, ExternE, between the European Commission and the US Department of Energy to quantify the damage costs imposed on society and the environment due to energy use (ExternE, 2005; e.g. Bickel and Friedrich, 2005). It follows a logical, stepwise progression from pollutant emissions to determination of impacts, and subsequently a quantification of damage costs in monetary terms (Figure 2.1).

Some pathways are fully characterised in a simple linear fashion as shown here. A good example concerns the quantification of the effects on human health of direct PM and precursor emissions, for which inhalation is the only relevant exposure route. In this case, it is necessary to quantify the pollutant emission, describe its dispersion and the extent to which the population is exposed, apply a concentration-response function, and finally evaluate the economic impact. Pathways for other pollutants may be significantly more complex.

Figure 2.2 illustrates the case for pollutants such as some heavy metals and organic compounds, where estimating total exposure may require information not just on exposure to pollutant concentrations in air, but also on consumption of various types of food and drinks. In these cases, it is possible that the inhalation dose may be only a small part of the total, as most impacts associated with exposure can occur through ingestion.

Figure 2.1 The impact pathway approach

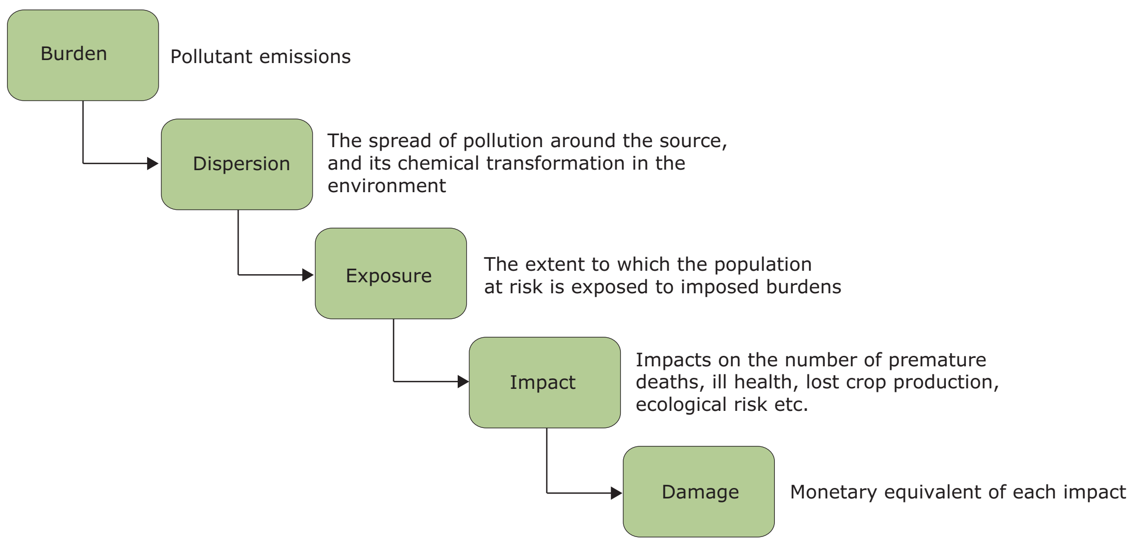
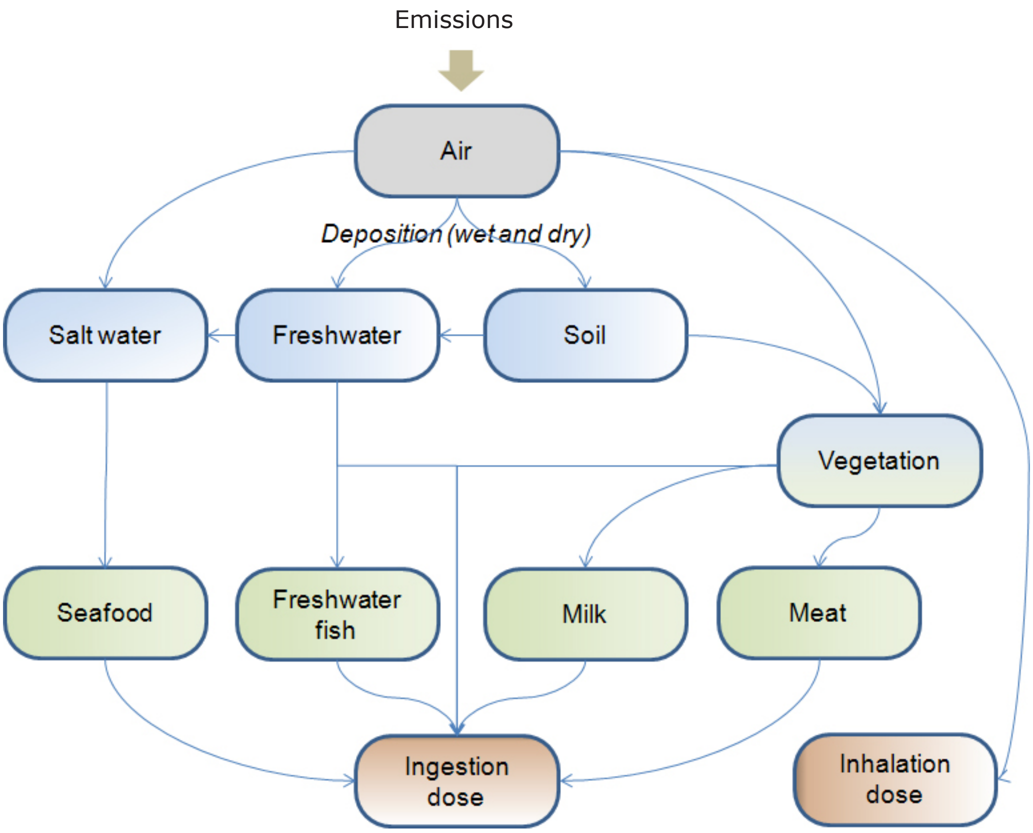


Figure 2.2 Pathways taken into account for estimating health impacts of air pollutants



2.2 E-PRTR emissions data

The damage costs determined in this report are based upon the emissions to air of selected pollutants reported by 14 325 individual facilities to the E-PRTR pollutant register for the years 2008 to 2012. The most recent version of the E-PRTR database available at the time of writing was used in the study (EEA, 2014c). The pollutants included in the assessment were:

- the main air pollutants: ammonia (NH_3), nitrogen oxides (NO_X), non-methane volatile organic compounds (NMVOCs), particulate matter (PM_{10}) and sulphur oxides (SO_X);
- heavy metals: arsenic, cadmium, chromium, lead, mercury and nickel;
- organic pollutants: benzene, dioxins and furans, and polycyclic aromatic hydrocarbons (PAHs ([6]);
- carbon dioxide (CO_2).

The E-PRTR register contains information on releases to air for 31 countries — 27 EU Member States and Iceland, Norway, Serbia and Switzerland. Country-specific damage costs (see Section 2.3) were not available for Iceland, and nor were any CO_2 emissions for Serbian facilities and so these two countries were not included in the analysis.

Prior to the estimation of damage costs, a number of E-PRTR data points were revised to correct apparent errors in the reported emissions (see Annex 1). These anomalies were identified either because there was one value amongst a time-series of emission values which differed by an order of magnitude or more, or because the reported value was a significant outlier while other pollutants reported from the same facility were consistent with the magnitude of emissions reported from other facilities.

There were also a number of instances where the official reporting of pollutants from facilities was identified as being potentially incomplete. In such instances, the evaluated damage costs

([6]) The derived damage costs for PAHs assume that PAH emissions are available as benzo-a-pyrene (BaP)-equivalents. In actuality, the E-PRTR Regulation (EU, 2006) requires emissions to be estimated for 4 PAH species, including BaP, on a mass basis.

are underestimated for the facility concerned. In particular the following instances were noted:

- Reporting of PM_{10} emissions, even from large facilities, appears very incomplete. For example, the facility estimated as having the highest aggregated damage costs 'TETs Maritsa Iztok 2' — Bulgaria has not reported PM_{10} emissions for any of the years 2008–2012. In a list of the top 100 facilities ranked by damage costs, 16 have not reported PM_{10} emissions, even though emissions above the E-PRTR thresholds might typically be expected for at least some of these facilities given the reported magnitude of emissions for other pollutants. In such cases the estimated damage costs are under-estimated, and such instances clearly bias the ranking of facilities against those facilities whose operators have been more conscientious in reporting complete data.

- A number of facilities have not reported emission estimates of any pollutants for certain years, despite it being likely such facilities did operate during these years. This includes for two facilities listed in the top 30 facilities ranked by damage costs, Longannet Power station (United Kingdom) for year 2011 and ILVA S.P.A. Stabilimento di Taranto (Italy) for 2012. Both these facilities would have ranked higher in the list of facilities having the highest damage costs had emissions data for the missing year been reported.

- The completeness of reporting of heavy metals and organic pollutants is generally poor.

No 'gap-filling' of missing emissions data was performed.

As described in Chapter 1, the E-PRTR provides information from specific industrial facilities. The E-PRTR Regulation (EU, 2006) defines the industrial sectors that must report information to the register. In addition, for this defined list of sectors, the Regulation includes reporting thresholds for both pollutants and activities. Facilities only have to report information to the register if their rate of activity exceeds the defined threshold and the emissions of a given pollutant exceed the pollutant-specific thresholds. In the first instance, E-PRTR compliance issues are the responsibility of the respective national competent authorities.

In practice, this means that many smaller facilities do not report emissions to E-PRTR, and all facilities regardless of their size need only report emissions of those pollutants that exceed the respective thresholds. The E-PRTR is therefore not designed to capture all emissions from industrial sectors.

Table 2.1 Comparison of the E-PRTR emissions data for 2012 with the corresponding national emission inventory total emissions

Pollutant	Emissions reported to E-PRTR (tonnes)	Aggregated national total emissions (tonnes)	% E-PRTR emissions of national totals
NH_3	194 183	3 714 680	5 %
NMVOC	457 536	6 860 090	7 %
NO_x	2 396 000	8 653 310	28 %
PM_{10}	131 164	1 885 176	7 %
SO_x	2 521 361	4 007 131	63 %
CO_2 ([a])	1 923 456 000	4 300 398 274	45 %
Arsenic	24.1	203.6	12 %
Cadmium	11.6	87.7	13 %
Chromium	77.3	359.2	22 %
Lead	310	2542	12 %
Mercury	28.1	77.8	36 %
Nickel	246	852	29 %
Benzene	2988	N.A. ([b])	-
PAHs	61.0	1041.2	6 %
Dioxins and furans	0.00072	0.00169	43 %

Note: ([a]) CO_2 reported to E-PRTR by facilities includes emissions from both fossil fuel and biomass. The value for the aggregated national total of CO_2 reported by countries to UNFCCC has thus had biomass CO_2 emissions added. These latter emissions are reported separately by countries, but are not included in the official national totals.

([b]) 'N.A.' denotes 'not available'.

To provide an illustration of the overall 'completeness' of the E-PRTR register, Table 2.1 provides a comparison of the 2012 emissions data for selected pollutants reported to E-PRTR, with the national total emissions for the same year reported in the national emission inventories submitted by countries to the UNECE LRTAP Convention (EEA, 2014a) and, for CO_2, under the EU Greenhouse Gas Monitoring Mechanism (EEA, 2014b) and UNFCCC. The emission inventory totals include emission estimates for those sectors not included in E-PRTR, such as small industrial sources as well as 'diffuse' sources such as transport and households. Sources such as these, not included in the E-PRTR, can make a very substantial contribution to the overall population exposure. With the exception of SO_2, Table 2.1 shows that for most pollutants, other sources not included in E-PRTR, produce the majority of emissions. Therefore, the damage costs estimated in this study clearly do not represent the total damage costs caused by air pollution across Europe.

2.3 General approach

It is possible to model the pollution impacts arising from specific industrial facilities in detail. The ExternE Project has undertaken this type of work extensively since the early 1990s (CIEMAT, 1999). However, such an analysis would be extremely resource intensive and costly if the aim were to model simultaneously and in detail the individual emissions, dispersion and impacts from the approximately 13 000 facilities covered by the E-PRTR. Some methodological simplification is thus necessary.

The simplified analysis used in this study applies the following approach:

1. Averaged country-specific damage costs per tonne of each pollutant were quantified;

2. Factors to account for any systematic variation in damage cost per tonne between the national average and specific sectors were developed (e.g. to account for typical differences in the location and height at which emissions from industrial sources are released, which will affect dispersion and hence exposure of people and ecosystems);

3. E-PRTR emissions data for each facility were multiplied by the national average damage cost per tonne estimates for each reported pollutant, with the sector-specific adjustment factors applied where available.

The main modelling work undertaken in this updated study addressed the first of these steps. A detailed description of the modelling undertaken to develop national average damage costs per tonne of pollutant is provided in Annex 2 (for the main air pollutants) and Annex 3 (for the heavy metals and organic pollutants).

For the main air pollutants NH_3, NO_X, NMVOCs, $PM_{2.5}$, and SO_2, the first step followed the approach described by Holland et al. (2005d) for developing updated marginal damage costs for the IPPC Directive reference document Economics and Cross Media Effects (European Commission, 2006a). The updated results in terms of damage cost per tonne of pollutant emission are different to those applied in the earlier report (EEA, 2011), as updated dispersion modelling from the EMEP model (EMEP, 2014) has been used in the present analysis, as well as adoption of recommendations from the recent World Health Organization's (WHO) HRAPIE study (WHO, 2013a), and refinement of some pollutant-specific effects e.g. for chronic bronchitis. Further methodological details are provided in Annex 2.

The second step — introduction of sector-specific factors — used information from the Eurodelta II study (Thunis et al., 2008). Eurodelta II compared air quality modelling results from a number of European-scale dispersion models, including an assessment of emission sources by sector. This enabled derivation of adjustment factors for four countries: France, Germany, Spain and the United Kingdom. For the present study, therefore, country-specific adjustment factors were applied to these four countries, and a sector-specific average value was used to make adjustments for the other countries. This requires that the E-PRTR facilities are mapped onto the sector descriptions used by Eurodelta II. Further details are provided in Annex 4.

The Eurodelta II analysis is subject to certain limitations, for example:

- the geographic domain of the models used does not cover the full area impacted by emissions from countries included in the E-PRTR;

- assumptions on stack height for the different sectors appear simplistic.

However, using the Eurodelta II national sector adjustment values in this report addresses the concern that a blanket application of national average data may overestimate the damage costs attributed to industrial facilities, if factors such as location and height at which emissions from industrial sources are typically released are not factored into the analysis.

In the final step — multiplying emissions data by the estimates of damage cost per tonne to quantify the total damage costs — PM_{10} data from the E-PRTR are converted to $PM_{2.5}$ by dividing by a general factor of 1.54. This conversion is necessary for consistency with the damage functions agreed under the CAFE programme and the dispersion modelling carried out by EMEP.

The follow sections describe in more detail the approaches used to determine the country-specific damage costs for the main air pollutants, heavy metals and organic pollutants, and CO_2. For the former two pollutant groups, additional methodological details are provided in the annexes to this report.

Damage costs throughout the report are expressed in 2005 euros.

Main air pollutants

Analysis of the impacts of the main air pollutant emissions (NH_3, NO_X, PM, SO_2 and NMVOC) addresses effects on human health, crops and building materials assessed against exposure to $PM_{2.5}$, ozone and acidity. The quantified health effects of SO_2, NO_X, NH_3 and NMVOCs result from the formation of secondary PM and ozone through chemical reactions in the atmosphere. The possibility of direct health effects occurring as a result of direct exposure to NO_X and SO_2 is not ruled out but such effects are assumed to be accounted for by quantifying the impacts of fine PM exposure. Quantifying them separately would therefore risk, at least to some extent, a double counting of their effects.

An important assumption in the analysis is that all types of particle of a given size fraction (e.g. $PM_{2.5}$ or PM_{10}) are equally harmful per unit mass. Alternative assumptions have been followed elsewhere (e.g. in the ExternE project) but here the approach used in the CAFE analysis was employed, following the recommendations of the Task Force on Health (TFH) coordinated by WHO Europe under the Convention on Long-range Transboundary Air Pollution (LRTAP Convention). This position is retained in the conclusions of the REVIHAAP Project assessing evidence on health aspects of air pollution (WHO, 2013b). Importantly for the purpose of the current report, it is noted that 'epidemiological studies continue to report associations between sulfates or nitrates and human health'.

This report does not quantify certain types of impact, for example ecosystem damage and harm to biodiversity caused by acidic and nitrogen deposition and exposure to ozone, and acid damage to cultural heritage such as cathedrals, other fine buildings and monuments. This should not be interpreted as implying that these effects are unimportant. Rather, they are not quantified because of a lack of data at some point in the impact pathway.

Included in the estimation of damage costs of the main air pollutants is an extensive list of health impacts, ranging from mortality to days with respiratory or other symptoms of ill health. In economic terms, the greatest effects concern exposure to primary and secondary PM leading to mortality, the development of chronic bronchitis in adults and days of restricted activity including work-loss days.

Recognising methods developed elsewhere, a sensitivity analysis has been performed using two commonly applied methods for valuing mortality — the mean value of statistical life (VSL) and the median value of a life year (VOLY) (Box 2.1) (OECD, 2012).

The debate about the correct approach to use for mortality valuation does not extend to the other pollutants considered here — heavy metals and organic pollutants. For these two pollutant groups, it is considered that exposure causes the onset of cancers or other forms of serious ill health that lead to a more substantial loss of life expectancy per case than for the main air pollutants and hence the use of the value of statistical life in these instances is considered appropriate.

The analysis of crop damage from exposure to ozone covers all of the main European crops. It does not, however, include an assessment of the effects on the production of livestock and related products such as milk. Material damage from the deposition of acidic or acidifying air pollutants was one of the great concerns of the acid rain debate of the 1970s and 1980s. This analysis accounts for the effects of SO_X emissions on a variety of materials, the most economically important being stone

Box 2.1 Approaches for estimating the health-related costs of air pollution

This report applies two contrasting but complementary approaches for valuing health damage caused by main air pollutants — the value of statistical life (VSL) and the value of a life year (VOLY).

- *The value of statistical life (VSL)* — an estimate of damage costs based on how much people are willing to pay for a reduction in their risk of dying from adverse health conditions;
- *The value of a life year (VOLY)* — an estimate of damage costs based upon the loss of life expectancy (expressed as potential years of life lost, or YOLLs). This measure takes into account the age at which deaths occur by giving greater weight to deaths at younger age and lower weight to deaths at older age.

Further details on the two methods, including a description of the health-related impacts which are included in the quantification methodologies, are provided in Annex 2.

and zinc/galvanised steel. Rates of damage have, however, declined significantly in Europe in recent decades in response to reduced emissions of SO_X, particularly in urban areas. Unfortunately, it is not yet possible to quantify the damage costs caused by air pollution's impact on monuments and buildings of cultural merit.

Further information on the methods used to quantify the effects of the main air pollutants is given in Annex 2.

Heavy metals and organic pollutants

As is the case for the major main air pollutants, assessment of the damage costs of heavy metals and organic pollutants is incomplete, particularly with respect to quantifying ecosystem damage costs. Direct analysis for these pollutants focuses on health effects, particularly cancers but also, for lead and mercury, neuro-toxic effects leading to IQ loss and subsequent loss of earnings potential. The RiskPoll model has been adopted for this part of the work (Spadaro and Rabl, 2004, 2008a, 2008b). Further details of this part of the analysis are given in Annex 3. The Annex contains information on a more extensive list of pollutants than those covered in this report, demonstrating that the methods can be extended beyond the current scope of work.

Where appropriate, the analysis takes account of the types of cancer identified for each pollutant in developing the impact pathways for each. Exposure only comprises inhalation where lung cancer is the only observed effect of a particular substance. For others, it is necessary to estimate total dose through consumption of food and drink as well as inhalation as shown in Figure 2.2. The valuation process takes account of the proportion of different types of cancer being fatal and non-fatal.

A complication arises because many of these pollutants are associated with PM upon release. By only taking account of their carcinogenic and neuro-toxic properties and ignoring their possible contribution to the other impacts of fine PM, it is possible that the total impact attributed to heavy metal and organic pollutant emissions is underestimated. However, quantifying the effects of PM and some effects of the trace pollutants separately may imply a risk of double counting, at least with respect to fatal cancers ([7]). This issue is discussed further in Chapter 4, where it is concluded that the overall effect of any double counting on the final results is very small, and that knowledge of the carcinogenic impact of these pollutants is useful.

Carbon dioxide

There is a variety of approaches used to estimate the economic costs associated with CO_2 emissions, and the benefits of mitigating emissions for policy assessment purposes. Such approaches are however recognised as being very uncertain. The Intergovernmental Panel on Climate Change (IPCC) has highlighted the significant difficulties and high uncertainties associated with approaches used for the economic assessment of climate change risks,

([7]) This does not apply to damage from neuro-toxic effects or the non-mortality costs of cancers related to healthcare, pain and suffering, and loss of productivity.

noting that there is a very wide range of values available in the literature, ranging between a few dollars and several hundreds of dollars per tonne of carbon in 2000 to 2015 (IPCC, 2014).

One type of approach, referred to as the social cost of carbon (SCC), attempts to estimate the long-term damage costs to society caused by the emission of one tonne of GHG (e.g. Stern, 2007; US EPA, 2013a). These approaches thus also potentially provide an indication of the current-day benefits arising from mitigating one tonne of emissions. However, SCC approaches are subject to well-documented limitations, with concerns raised as to what extent they capture the uncertainties related to future societal changes, future emissions and related future temperature changes and impacts. Furthermore, the nature and scale of future impacts, economic discount rates, how to monetise certain impacts such as biodiversity loss and how to correctly reflect in these approaches events of low probability but high-impact, provide additional significant sources of uncertainty.

Given the difficulty on deciding an appropriate value for the present day damage costs per tonne of emitted CO_2, other approaches have been developed based on the cost to society of avoiding dangerous levels of climate change. The so-called target-consistent approach, starts from a GHG reduction target, to be achieved in order to be consistent with a long term climate target. An associated carbon value that would result in the necessary emission reductions is subsequently derived. This type of approach has for example been used since 2009 for carbon valuation by the United Kingdom's Department of Energy and Climate (DECC, 2009; DECC, 2011; DECC, 2013).

The previous EEA report (EEA, 2011) applied such a target-consistent approach, based on the marginal abatement costs used at that time by the UK government for carbon valuation in public policy appraisal (DECC, 2011) — a price of EUR_{2005} 33.6 per tonne was used. While this figure reflected the views of the UK government rather than any consensus-based estimate, it was considered reasonably representative and consistent with other available approaches, either in relation to damage or abatement costs.

The present report uses an updated approach based upon recent modelled ETS carbon price forecasts performed for the European Commission to support the proposal for a 2030 climate and energy policy framework (European Commission, 2014). While the 2011 report used only a single value to estimate CO_2-related damage costs, the present report applies a range of values. This approach provides a reflection of the costs associated with decreasing CO_2 emissions in the EU over time in line with the required reduction necessary to meet the current policy objective of limiting future limit average global surface temperature increase to two-degrees. The range of values used is:

- A lower value of EUR_{2005} 9.5 per tonne CO_2 (based upon a value of EUR 10 per tonne CO_2 in 2010 prices ([8])) reflecting the modelled ETS price in 2020 based on a reference scenario (implementation of current legislation);

- A higher value of EUR_{2005} 38.1 per tonne CO_2 (based upon a value of EUR 40 per tonne CO_2 in 2010 prices) reflecting the projected carbon price in 2030 in a central scenario of 40% domestic GHG emission reduction by 2030 compared to 1990 (European Commission, 2014).

Regardless of the value or type of methodological approach being used to estimate greenhouse gas related damage costs, it is however clear that work needs to continue on the better estimation of the economic impacts of greenhouse gas emissions on society.

([8]) Damage cost per tonne values for CO_2 were converted to 2005 prices using Eurostat GDP data (http://epp.eurostat.ec.europa.eu/portal/page/portal/national_accounts/data/database) accessed 1 June 2014.

3 Results

The results of this work are described in the following several parts.

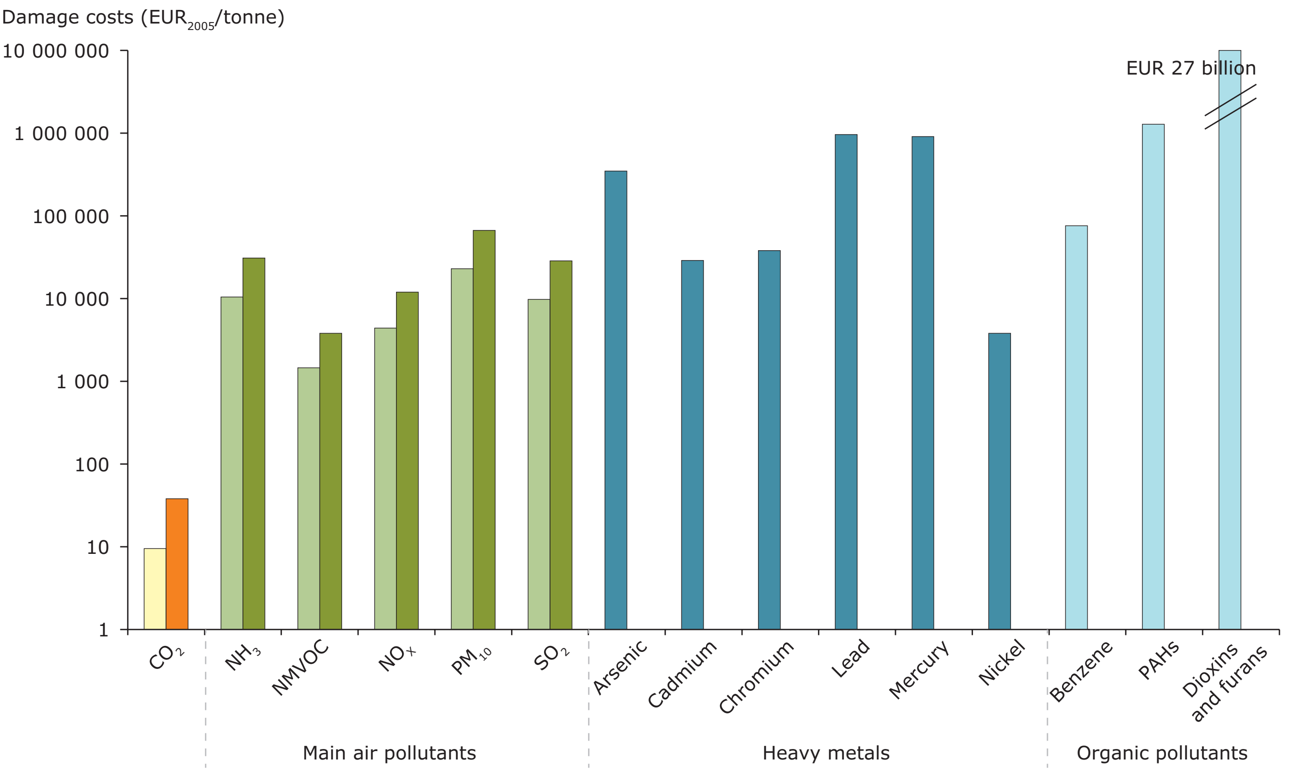
- The first section, Section 3.1, presents the updated damage costs per tonne of emission determined for each of the selected pollutants. These results serve to link emissions and the final damage cost estimates presented subsequently. A comparison of the updated results calculated in this work with those from the previous EEA report (EEA, 2011) is provided.
- Section 3.2 highlights the importance of taking into account differences of (fuel) efficiency between facilities by exploring several proxy methods of normalising damage costs.
- Section 3.3 presents a short assessment of potential damage cost savings based on a recent EEA report that assessed the hypothetical emission reduction potential of NO_X, SO_2 and dust from more than 1 500 of Europe's large combustion plants operating in 2009 (EEA, 2013b). The present report draws on the results from that study to illustrate the scale of potential benefits, in terms of reduced damage costs, that would occur were certain facilities to reduce future levels of emissions.

3.1 Revealing the costs of air pollution from E-PRTR facilities

Updated damage cost estimates

Figure 3.1 and Table 3.1 show the variation in the updated damage costs per unit of emission between pollutants. For illustrative purposes, data have been

Figure 3.1 Estimates of the European average damage cost per tonne emitted for selected air pollutants (2005 prices)



Note: Note the logarithmic scale on the Y-axis.
For CO_2, the range corresponds to the selected lower and higher values used. For the main air pollutants, the lower value of the range shows the average valuation of mortality calculated using the VOLY approach, with the upper corresponding to the average VSL approach for mortality valuation.

Table 3.1 Estimates of the European average damage cost per tonne emitted for selected air pollutants (2005 prices)

Pollutant	Average damage cost (EUR_{2005} per tonne)		
		Low VOLY for the main air pollutants	High VSL for the main air pollutants
CO_2 low	9.5	-	-
CO_2 high	38.1	-	-
NH_3	-	10 460	30 908
NMVOC	-	1 461	3 808
NO_X	-	4 419	11 966
PM_{10}	-	22 990	66 699
SO_2	-	9 792	28 576
Arsenic	349 000	-	-
Cadmium	29 000	-	-
Chromium	38 000	-	-
Lead	965 000	-	-
Mercury	910 000	-	-
Nickel	3 800	-	-
Benzene	76 000	-	-
PAHs	1 279 000	-	-
Dioxins and furans	27 000 000 000	-	-

averaged across countries (i.e. for all of the selected pollutants except CO_2, lead and mercury). Results for each country are provided in Annexes 1 and 2.

As illustrated in the previous EEA report (EEA, 2011), Figure 3.1 and Table 3.1 show that the values for damage cost per tonne emitted vary substantially between pollutants, with nine orders of magnitude difference between the values for CO_2 and dioxins. There is an approximate ordering of the different pollutant groups, with the organic pollutants the most hazardous per unit of emission, followed by the heavy metals, main air pollutants, and finally CO_2.

The country-specific estimated damage costs per unit of emission provided in Annexes 1 and 2 again vary significantly among emitting countries for various reasons, including the following:

- The density of receptors (people, ecosystems) varies significantly around Europe e.g. the regional population density.
- Pollutant dispersion patterns and differences in atmospheric chemistry (such as chemical transformation rates) which are dependent upon the location of emissions.
- Some emissions disperse out to sea and do not affect life on land, an issue clearly more prominent for countries with extensive coastlines such as the United Kingdom or Ireland compared to landlocked countries such as Austria or Hungary.

For some pollutants the site of release is relatively unimportant in determining the magnitude of damage costs. The pollutants CO_2 and mercury are good examples, although their impacts differ greatly.

Figure 3.2 Illustration of variation in national average damage costs per tonne of NO_x emissions

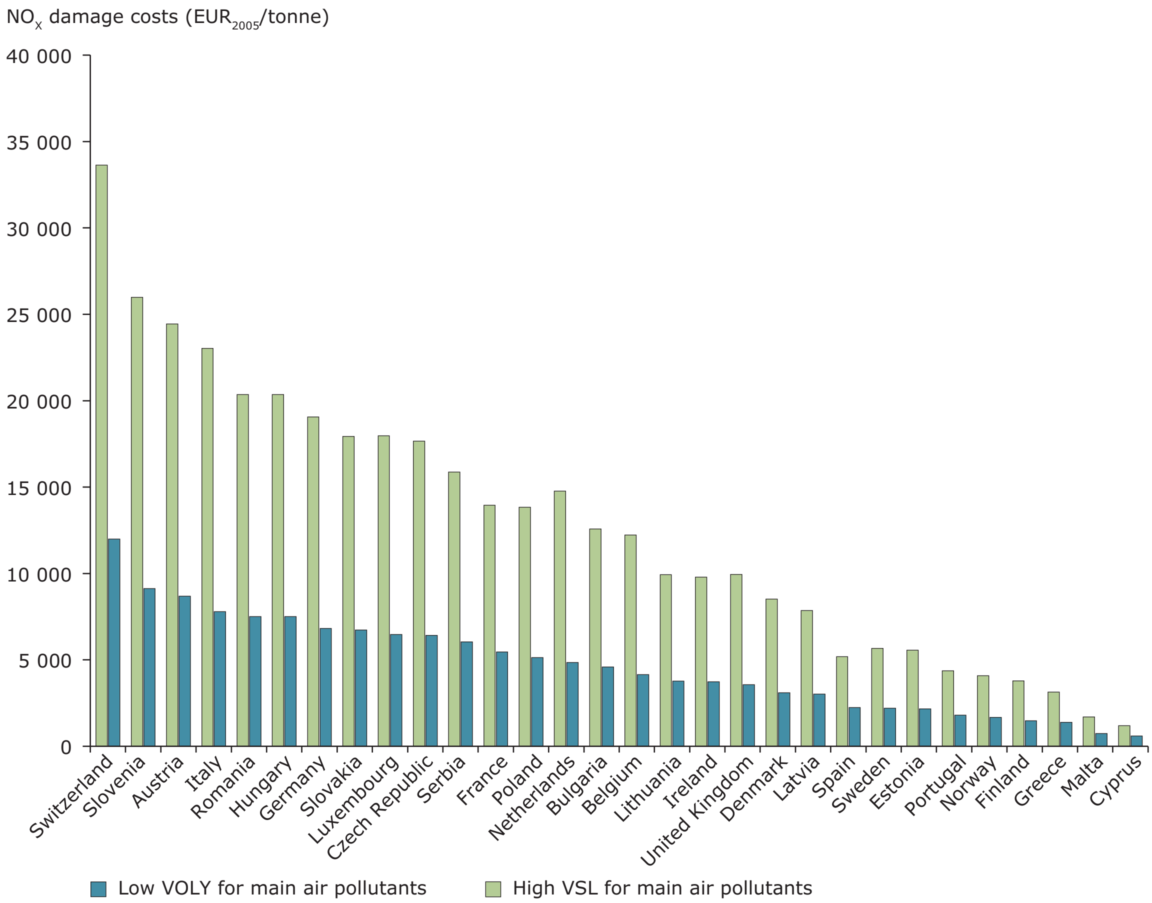
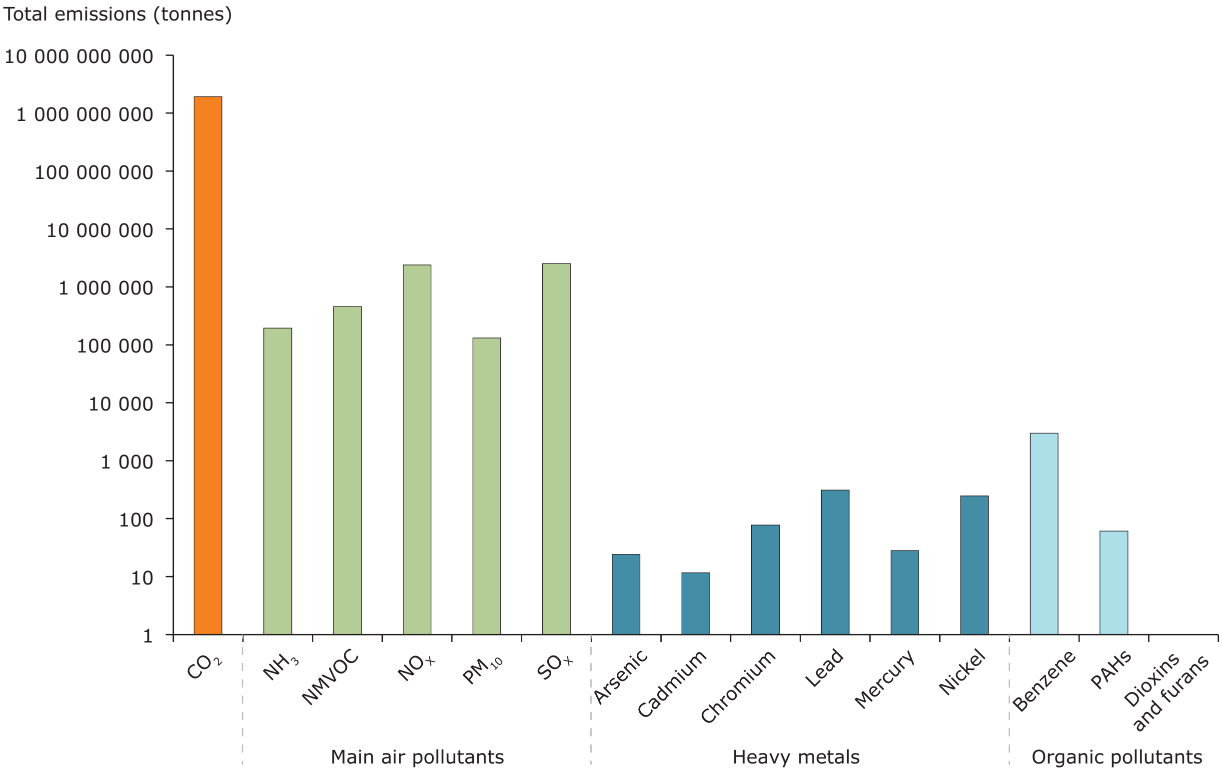


Figure 3.2 illustrates the variation in the average damage costs attributed to NO_x for each country. There is a factor 20 difference between the country with the lowest damage cost per tonne (Cyprus) and the highest (Switzerland). Figure 3.2 also shows the sensitivity of results to the methods used for valuing mortality (i.e. VSL > VOLY).

Using the country-specific damage costs per unit of emissions, the damage costs caused by each facility reported under the E-PRTR may be quantified by multiplying the emissions of the selected pollutant from each facility by the respective damage cost per tonne for each pollutant, corrected where appropriate to account for differences between sectors. Aggregated results, and those for the individual facilities, are shown in the following sections.

Total emissions of each pollutant from the E-PRTR are shown in Figure 3.3 for 2012, one of the years addressed in this report. The emissions of differing pollutants vary in scale by twelve orders of magnitude. Emissions are dominated by CO_2, followed by the main air pollutants and heavy metals. Reported emissions of organic pollutants are so small (under 1 kg for dioxins) they are not visible on the graph. The ordering of pollutants by emissions is roughly the reverse of the ordering by the damage cost per tonne, as shown in Figure 3.1. Thus, those pollutants that are the most hazardous per unit of emission tend to be emitted in the smallest quantities.

Figure 3.3 Emissions to air of selected pollutants from E-PRTR in 2012



Note: Note the use of a logarithmic scale on the Y-axis.

Aggregated damage costs

Aggregated damage costs for 2008 to 2012 are provided in Table 3.2 and illustrated in Figure 3.4. In Table 3.2, the range shown for CO_2 damage costs reflects the difference between the minimum and maximum selected values for carbon valuation. For the main air pollutants, the lower value of the range provided corresponds to the valuation of mortality calculated using the VOLY approach, whilst the upper value corresponds to cases where the VSL approach has been applied to mortality valuation. Other sources of uncertainty are not considered.

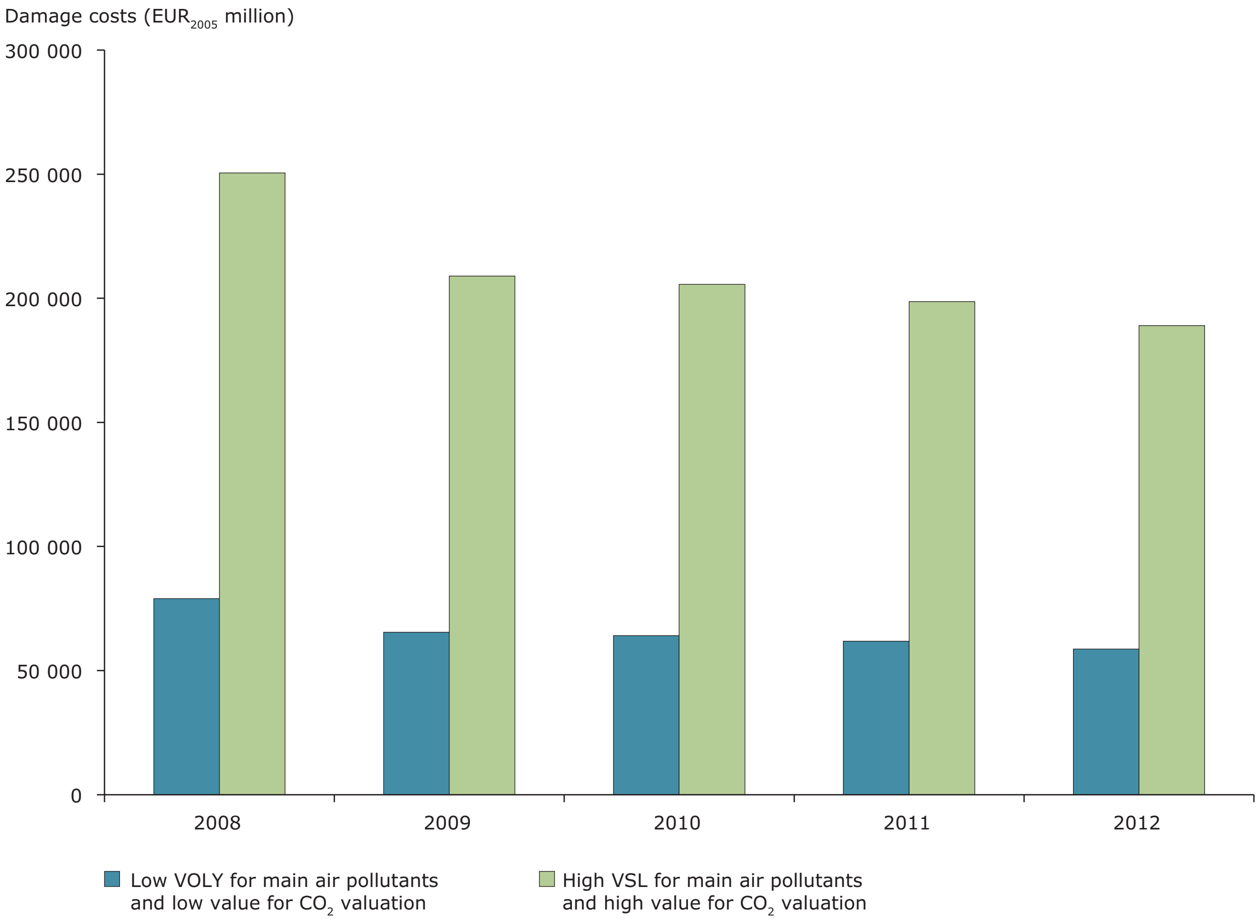
The aggregated cost between 2008 and 2012 of damage caused by emissions from E-PRTR industrial facilities was estimated as being at least EUR_{2005} 329 billion (and up to EUR_{2005} 1 053 billion) (Table 3.2). Damage costs caused by emissions from E-PRTR facilities declined in the years following 2008. Various contributory factors will have contributed to this decrease, including both the ongoing impacts of legislation and the economic recession in Europe which resulted in lower rates of industrial activity in years immediately after 2008. The wide range in the estimated damage costs in each year illustrates the large sensitivity of results in terms of both the values and methods used to calculate the pollutant-specific damage costs.

Table 3.2 Estimated damage costs aggregated by pollutant group, 2008–2012 (2005 prices)

Pollutant group	Aggregated damage cost (billion EUR_{2005})				
	2008	2009	2010	2011	2012
Main air pollutants (NH_3, NO_X, PM_{10}, SO_2, NMVOCs)	58–168	47–136	44–129	43–124	40–115
CO_2	20–82	18–73	19–76	18–74	18–73
Heavy metals (As, Cd, Cr, Hg, Ni, Pb)	0.53	0.34	0.43	0.34	0.34
Organic pollutants (benzene, dioxins and furans, PAHs)	0.22	0.11	0.17	0.22	0.10
Sum	**79–251**	**65–209**	**64–206**	**62–199**	**59–189**

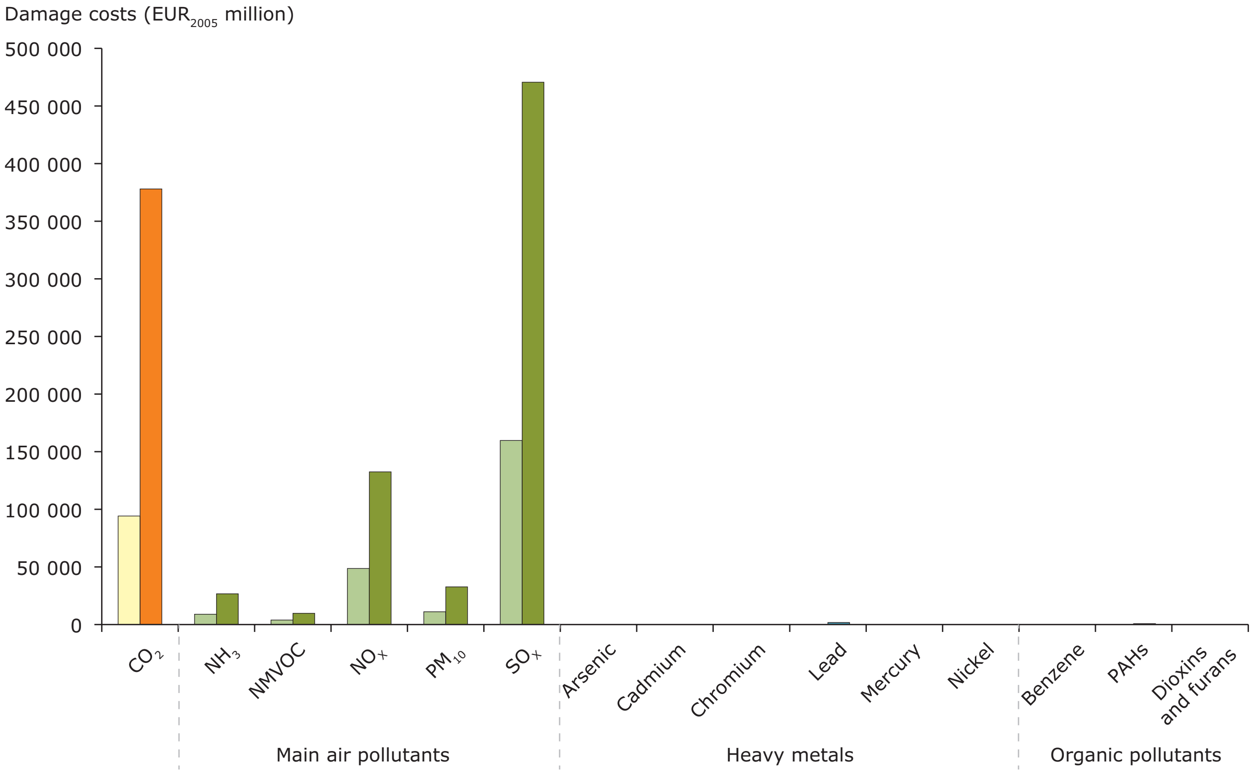
Note: For carbon valuation, the difference between the low and high estimates shown for each year reflects (i) the difference between the minimum (EUR_{2005} 9.5 per tonne CO_2) and maximum values (EUR_{2005} 38.1 per tonne CO_2). For the main air pollutants, the lower value of the range is a calculation of the valuation of mortality using the VOLY approach, whilst the upper value is a calculation using the VSL approach.

Figure 3.4 Estimated damage costs, 2008–2012



Note: The low-high range shows the differing results derived from the alternative approaches to a) mortality valuation for the main air pollutants and b) the difference between minimum (EUR_{2005} 9.5 per tonne CO_2) and maximum values (EUR_{2005} 38.1 per tonne CO_2) used in this report for carbon valuation.

Figure 3.5 Aggregated damage costs by pollutant, 2008–2012

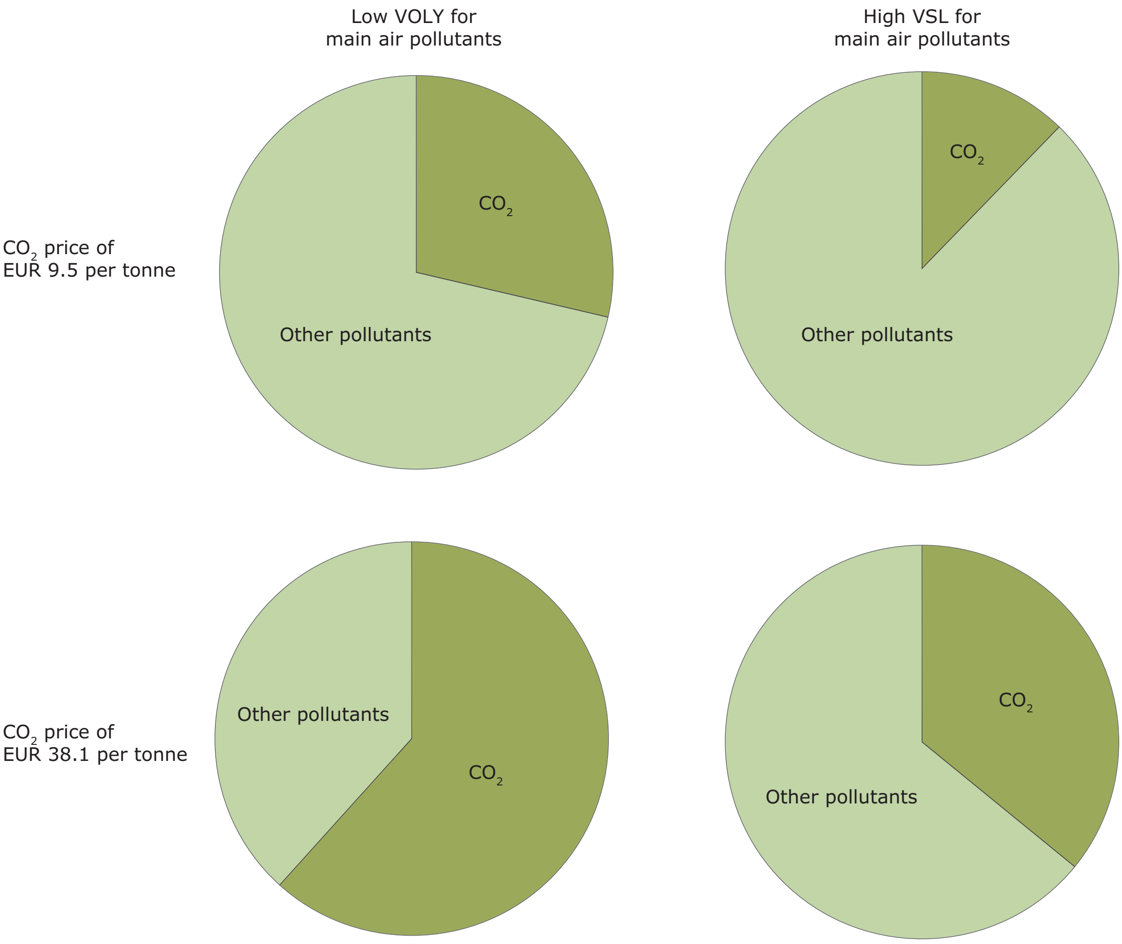


Note: The low-high ranges show the differing results derived from the alternative approaches to a) mortality valuation for the main air pollutants and b) the difference between minimum (EUR_{2005} 9.5 per tonne CO_2) and maximum values (EUR_{2005} 38.1 per tonne CO_2) used in this report for carbon valuation.

Figure 3.5 shows the aggregated damage costs by pollutant for 2008 to 2012. Damage costs are highest for the main air pollutants (SO_2, NO_X, PM_{10}, NH_3 and NMVOC) and CO_2. Quantified damage costs from the metals and organics are very small relative to the other pollutants, but still contribute hundreds of millions of euros harm to health and the environment, and at the local scale can cause significant adverse impacts.

The share of total estimated damage costs made by the different pollutants varies significantly depending upon the respective approaches used to estimate damage costs for the main air pollutants and CO_2. Figure 3.6 shows the changing contribution of these pollutants as a fraction of the total damage costs estimated. The proportion of CO_2 as a fraction of the total estimated damage costs varies widely depending on the approach used for CO_2 valuation, ranging from 12 % to 62 % of total damage costs.

Figure 3.6 Contribution of main air pollutants and CO_2 to total damage costs — impacts of differences in valuation methods and assumptions



Comparison with previous results

The results provided in this report using the updated methodology for the determination of damage costs of the main air pollutants are overall similar to those determined in the previous EEA report (EEA, 2011), in which damage costs for the year 2009 were determined to be at least EUR_{2005} 102–169 billion. The present study estimates the damage costs in that year as EUR_{2005} 111–200 billion when applying the same CO_2 value (EUR 33.6 per tonne) as used in the previous study. This is an increase in the estimated damage costs of 9 % based on the lower VOLY and 19 % for the higher VSL approaches for mortality valuation.

The main differences in the damage costs are due to the updated methodology for the main air pollutants (Table 3.3). Compared with the previous estimated pollutant damage costs (EEA, 2011), the updated modelling produces results (in terms of pollution-related damage) that are generally lower for NO_X and NH_3, slightly higher for PM_{10}, and significantly higher for NMVOCs and SO_2. The increase for NMVOCs is due largely to the new inclusion of secondary organic aerosols into the modelling, with the increase for SO_2 being attributed to changes in the emissions scenario used (now for year 2010) which has affected the reactivity of atmospheric sulphur as predicted by the EMEP model.

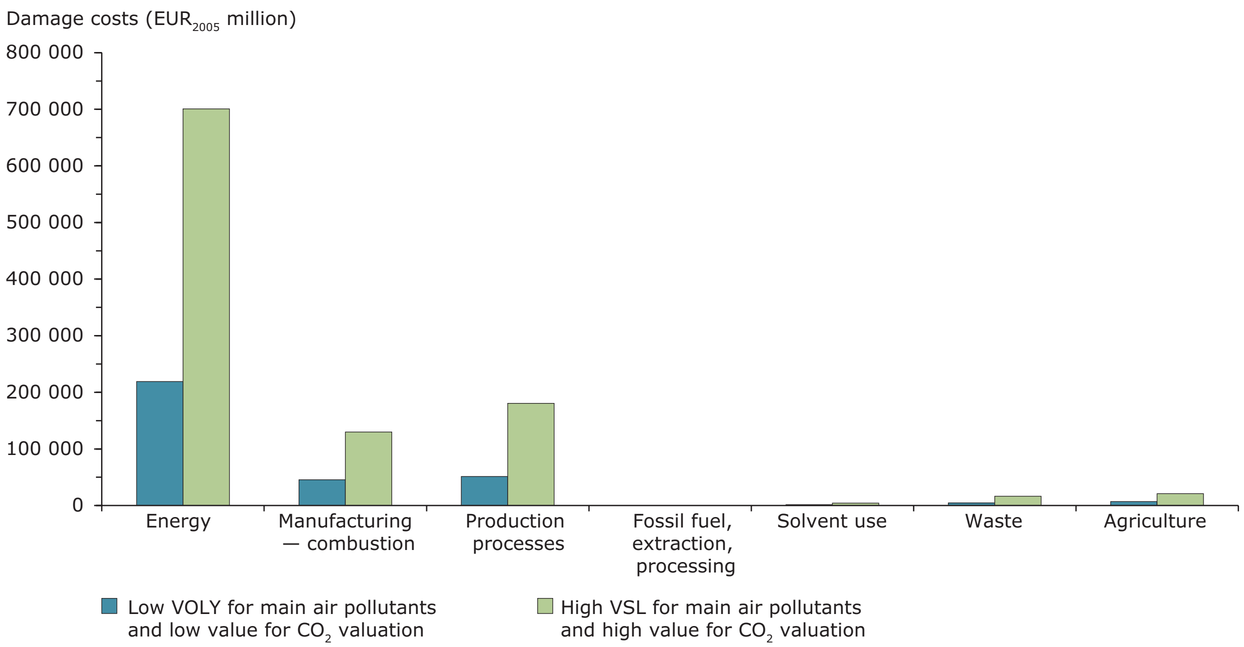
Table 3.3 also shows some small changes in the estimated damage costs for pollutants other than the main air pollutants, i.e. for those where the methodology has not been updated. These changes are due to corrections made by Member States to the reported emissions for 2009 in the E-PRTR after the publication of the previous study.

Table 3.3 Comparison of the damage costs for 2009 estimated in the previous 2011 EEA report and the present assessment (EUR$_{2005}$ million)

Pollutant	EEA 2011 report		Present report *	
	VOLY for main air pollutants	VSL for main air pollutants	VOLY for main air pollutants	VSL for main air pollutants
CO_2	63 230	63 230	63 976	63 976
NH_3	2 012	5 633	1 766	5 290
NMVOC	384	821	781	1 992
NO_x	14 584	39 437	9 673	26 393
PM_{10}	1 512	4 232	2 051	6 070
SO_x	19 974	54 912	32 602	96 261
Arsenic	10	10	8.5	8.5
Cadmium	0.3	0.3	0.3	0.3
Chromium	3.0	3.0	3.0	3.0
Lead	304	304	303	303
Mercury	28	28	28	28
Nickel	0.9	0.9	0.9	0.9
Benzene	0.3	0.3	0.3	0.3
PAHs	109	109	92	92
Dioxins and furans	23	23	21	21
Total	**102 174**	**168 743**	**111 306**	**200 437**

Note: * To allow comparison with the 2011 EEA report, a CO_2 price of EUR$_{2005}$ 33.6 per tonne is applied.

Figure 3.7 Damage costs for 2008–2012 aggregated by sector



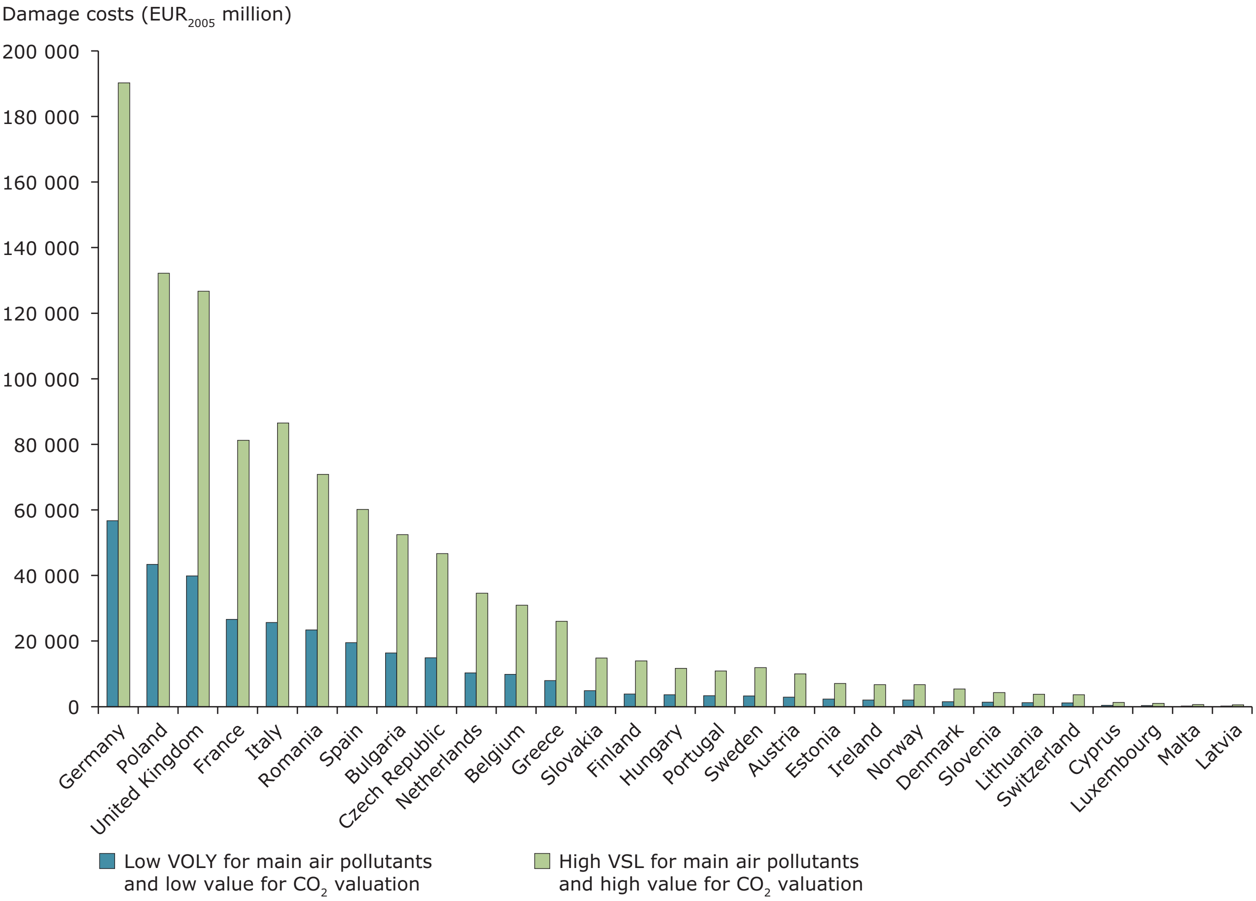
Note: The low-high range shows the differing results derived from the alternative approaches to a) mortality valuation for the main air pollutants and b) the difference between minimum (EUR$_{2005}$ 9.5 per tonne CO_2) and maximum values (EUR$_{2005}$ 38.1 per tonne CO_2) used in this report for carbon valuation.

Aggregated damage costs by sector and country

Of the industrial sectors included in the E-PRTR pollutant register, emissions from the energy sector contribute the largest share of the damage costs across the five-year period assessed (estimated as at least EUR_{2005} 219 billion (and up to EUR_{2005} 701 billion) (Figure 3.7). Sectors involving production processes and combustion used in manufacturing are responsible for most of the remaining estimated damage costs. Annex 4 provides a generic description of the mapping of E-PRTR sectors to the sector classifications used in this report.

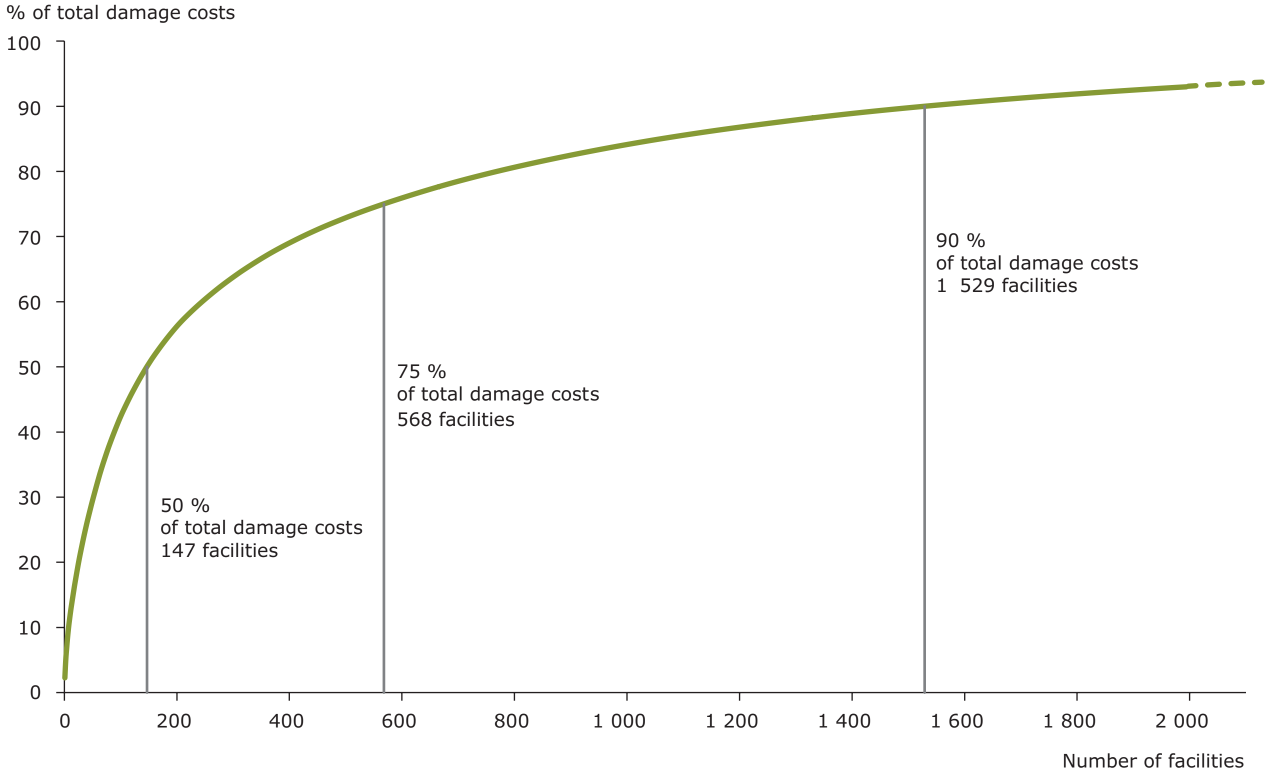
Aggregated results by country are presented in Figure 3.8. The highest aggregated damage costs are not surprisingly attributed to the larger countries and those with more polluting facilities especially power generating facilities included in the energy sector. The ordering is again very similar to that in the previous EEA report. Countries such as Germany, Poland, the United Kingdom, France and Italy, which have many large facilities, contribute the most to total estimated damage costs.

Figure 3.8 Aggregated damage costs by country, 2008–2012



Note: The low-high range shows the differing results derived from the alternative approaches to a) mortality valuation for the main air pollutants and b) the difference between minimum (EUR_{2005} 9.5 per tonne CO_2) and maximum values (EUR_{2005} 38.1 per tonne CO_2) used in this report for carbon valuation.

Figure 3.9 Cumulative distribution of the estimated damage costs associated with emissions of selected pollutants from E-PRTR facilities, 2008–2012



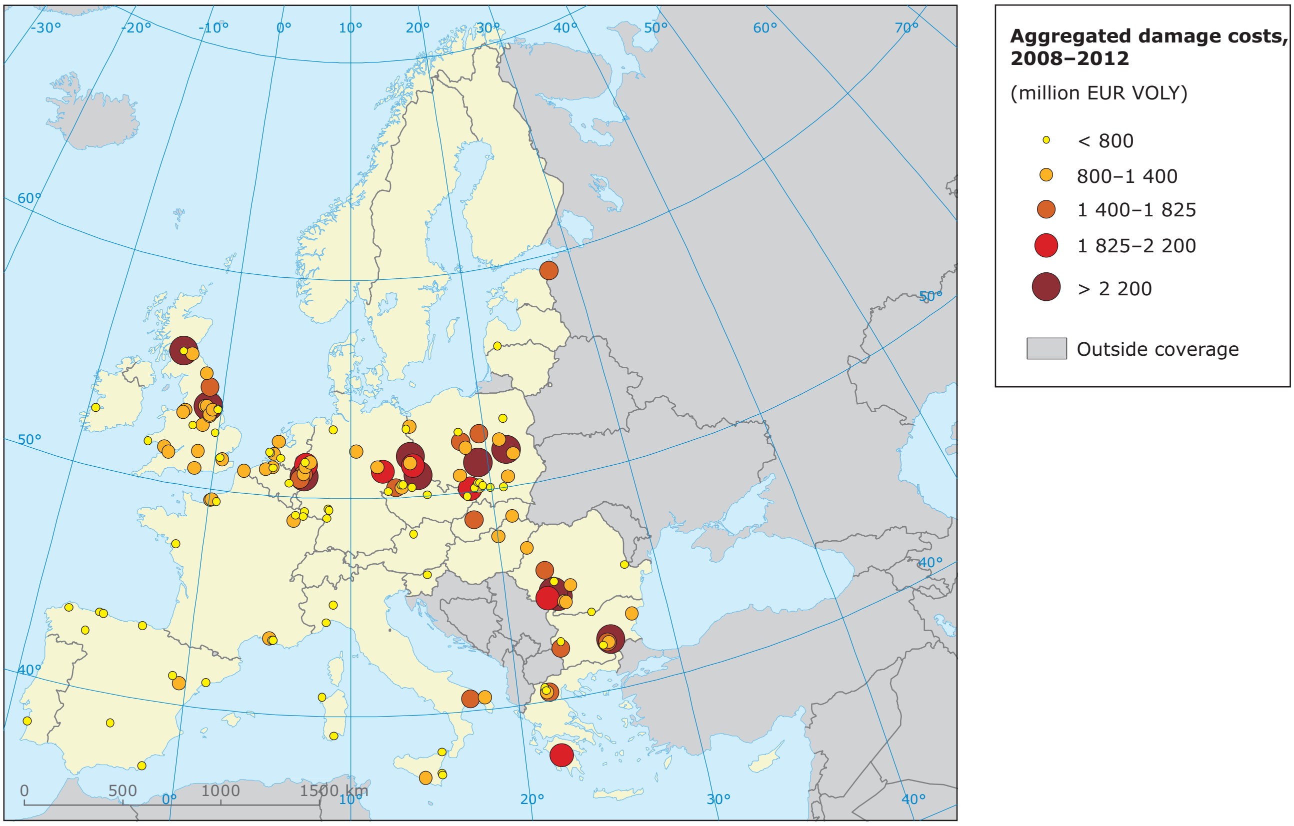
Note: The distribution is based on the lower VOLY approach for the main air pollutants and a CO_2 price of EUR_{2005} 9.5 per tonne.

Damage cost estimates for individual E-PRTR facilities

Figure 3.9 shows the cumulative distribution of the estimated damage costs for the E-PRTR facilities for the five years 2008–2012. As was observed in the previous EEA report (EEA, 2011) it remains clear that a very small number of individual facilities cause the majority of the damage costs. Fifty per cent of the total damage cost occurs as a result of emissions from just 147 (or 1 %) of the 14 325 facilities that reported data for releases to air during this period. Three quarters of the total damage costs were caused by the emissions of 568 facilities (4 % of the total number of facilities), and 90 % of damage costs are attributed to 1 529 facilities (11 % of the total). Map 3.1 shows the geographical distribution of the 147 facilities which contributed 50 % of the total damage costs. These findings should not, however, take focus away from the need to also regulate emissions from smaller facilities, which on the local scale can contribute significantly to air pollution and its subsequent harmful impacts.

Table 3.4 lists the 30 facilities estimated to cause the greatest aggregated damage costs for the selected pollutants across the five years covered in this study (2008–2012). Of these, 26 are power-generating facilities, mainly fuelled by coal/lignite and located predominantly in Germany and Eastern Europe.

Map 3.1 Location of the 147 E-PRTR facilities that contributed 50 % of the total damage costs estimated for 2008–2012



Note: The lower VOLY approach for the main air pollutants and a CO_2 price of EUR_{2005} 9.5 per tonne are applied. Of these facilities, 94 of the 147 facilities are categorised in E-PRTR as being power generating facilities.

Eight of the top 30 facilities are located in Germany; six are in Poland; four are in Romania; three are in Bulgaria and the United Kingdom, two are located in Greece; and one is located in each of the Czech Republic, Estonia, Italy and Slovakia.

It is also clear that the Top 30 facilities do not always appear to be reporting complete emissions data to E-PRTR. For example, the Bulgarian facility (TETs Maritsa Iztok 2, EAD) ranked first in terms of its overall damage costs, has not reported PM_{10} for any of the years 2008–2012 despite emissions of this pollutant being expected from a large facility of this nature. In a list of the top 100 facilities ranked by damage costs, 16 have not reported PM_{10} emissions, even though emissions above the E-PRTR thresholds might typically be expected for at least some of these facilities given the reported magnitude of emissions for other pollutants. Of the top 30 facilities ranked by damage costs, the Longannet power station (United Kingdom — number 10) did not report any pollutant emissions in 2011, while similarly the ILVA S.P.A. Stabilimento di Taranto facility (Italy — number 29) did not report any emissions data for 2012. The estimated damage costs from these facilities are therefore underestimated, and both would have ranked higher in the ranked list of facilities having damage costs had they reported emissions data for the missing years. Omissions such as these will cause inaccuracies, with any ranking of facilities biased against those whose operators have been more conscientious in reporting complete data.

Table 3.4 The top 30 E-PRTR facilities having the highest absolute damage costs from emissions of selected pollutants to air, 2008–2012

Number	Facility name	City	Country	Activity	Aggregated damage cost 2008–2012 (EUR$_{2005}$ million)	
					VOLY low	VSL high
1	'TETs Maritsa Iztok 2' EAD	Kovachevo	Bulgaria	Thermal power station	7 465	22 394
2	PGE Górnictwo i Energetyka Konwencjonalna S.A., Oddział Elektrownia Bełchatów	Rogowiec	Poland	Thermal power station	5 997	14 126
3	Sucursala Electrocentrale Turceni	Turceni	Romania	Thermal power station	4 916	13 761
4	Vattenfall Europe Generation AG Kraftwerk Jänschwalde	Peitz	Germany	Thermal power station	3 498	8 165
5	Drax Power Limited	Selby	United Kingdom	Thermal power station	3 482	8 039
6	Sucursala Electrocentrale Rovinari	Rovinari	Romania	Thermal power station	3 198	8 844
7	PGE Górnictwo i Energetyka Konwencjonalna S.A., Oddział Elektrownia Turów	Bogatynia	Poland	Thermal power station	2 797	6 925
8	Elektrownia 'Kozienice' S.A.	Świerże Górne	Poland	Thermal power station	2 667	6 580
9	RWE Power AG Kraftwerk Niederaußem	Bergheim	Germany	Thermal power station	2 276	4 172
10	Longannet Power Station	Kincardine	United Kingdom	Thermal power station	2 226	5 761
11	Regia Autonoma Pentru Activitati Nucleare — Sucursala Romag Termo	Drobeta Turnu Severin	Romania	Thermal power station	2 117	6 022
12	ThyssenKrupp Steel Europe AG Werk Schwelgern	Duisburg	Germany	Iron and steel production	2 048	5 316
13	PPC S.A. SES Megalopolis A'	Megalopoli	Greece	Thermal power station	1 872	5 103
14	EDF Rybnik S.A.	Rybnik	Poland	Thermal power station	1 870	4 574
15	Vattenfall Europe Generation AG Kraftwerk Lippendorf	Böhlen	Germany	Thermal power station	1 832	4 368
16	Kraftwerk Boxberg	Boxberg	Germany	Thermal power station	1 829	3 976
17	SC Electrocentrale Deva SA	Mintia	Romania	Thermal power station	1 819	5 066
18	Slovenské elektrárne a.s. — Elektrárne Nováky, závod	Zemianske Kostoľany	Slovakia	Thermal power station	1 814	5 003
19	Elektrárny Prunéřov	Kadaň	Czech Republic	Thermal power station	1 690	4 063
20	RWE Power AG Kraftwerk Neurath	Grevenbroich	Germany	Thermal power station	1 670	2975
21	Zespól Elektrowni Pątnów-Adamów-Konin S.A., Elektrownia Pątnów	Konin	Poland	Thermal power station	1 652	4 146
22	RWE Power AG	Eschweiler	Germany	Thermal power station	1 639	2 952
23	TETs 'Bobov dol'	Golemo selo	Bulgaria	Thermal power station	1 629	4 900
24	Eesti Energia Narva Elektrijaamad AS	Auvere küla, Vaivara vald	Estonia	Thermal power station	1 599	3 627
25	Polski Koncern Naftowy ORLEN S.A.	Płock	Poland	Refinery	1 586	3 944

Table 3.4 The top 30 E-PRTR facilities having the highest absolute damage costs from emissions of selected pollutants to air, 2008–2012 (cont.)

Number	Facility name	City	Country	Activity	Aggregated damage cost 2008–2012 (EUR$_{2005}$ million)	
					VOLY low	VSL high
26	PPC S.A. SES Agioy Dhmhtrioy	Agios Dimitrios, Ellispontos	Greece	Thermal power station	1 524	3 118
27	Teesside Integrated Iron and Steelworks	Redcar	United Kingdom	Iron and steel production	1 494	3 937
28	TPP 'Brikel'	Galabovo	Bulgaria	Thermal power station	1 430	4 386
29	ILVA S.P.A. Stabilimento di Taranto	Taranto	Italy	Iron and steel production	1 416	3 617
30	RWE Power AG Kraftwerk Frimmersdorf	Grevenbroich	Germany	Thermal power station	1 385	2 709

Note: A CO_2 price of EUR$_{2005}$ 9.5 per tonne is applied. The order of ranking is based on the lower VOLY approach.

3.2 Recognising relative differences in efficiency and productivity

Clearly, ranking facilities according to their aggregate damage costs provides little indication of the efficiency of production at a facility. It is clear that certain facilities have high damage cost estimates simply because of their size and high levels of production or activity. One large facility could pollute less than several smaller ones that generate the same level of service or output. Equally, the converse could also be true.

One weakness of the E-PRTR for assessment purposes is the lack of fuel consumption or production data for individual facilities, making it impossible to assess directly a facility's environmental impacts relative to its output. This report seeks to get around this shortcoming and illustrate the potential differences in facility efficiency by exploring several proxy methods of normalising damage costs to take into account the differences of efficiency between facilities.

Figure 3.8 showed the aggregated damage costs per country. An alternative way to rank countries is to normalise the estimated damage costs by introducing the concept of efficiency into the analysis. Normalising the damage costs by gross domestic product (GDP) to reflect the output of national economies results in significant changes in the ordering of countries. Certain countries previously shown as having the highest damage costs — Germany, the United Kingdom, France and Italy — drop significantly down the ranking, while Bulgaria, Romania and Estonia rise to the top (Figure 3.10). Poland remains toward the top of the rankings, reflecting the high amounts of pollutants at Polish facilities emitted relative to national gross domestic product.

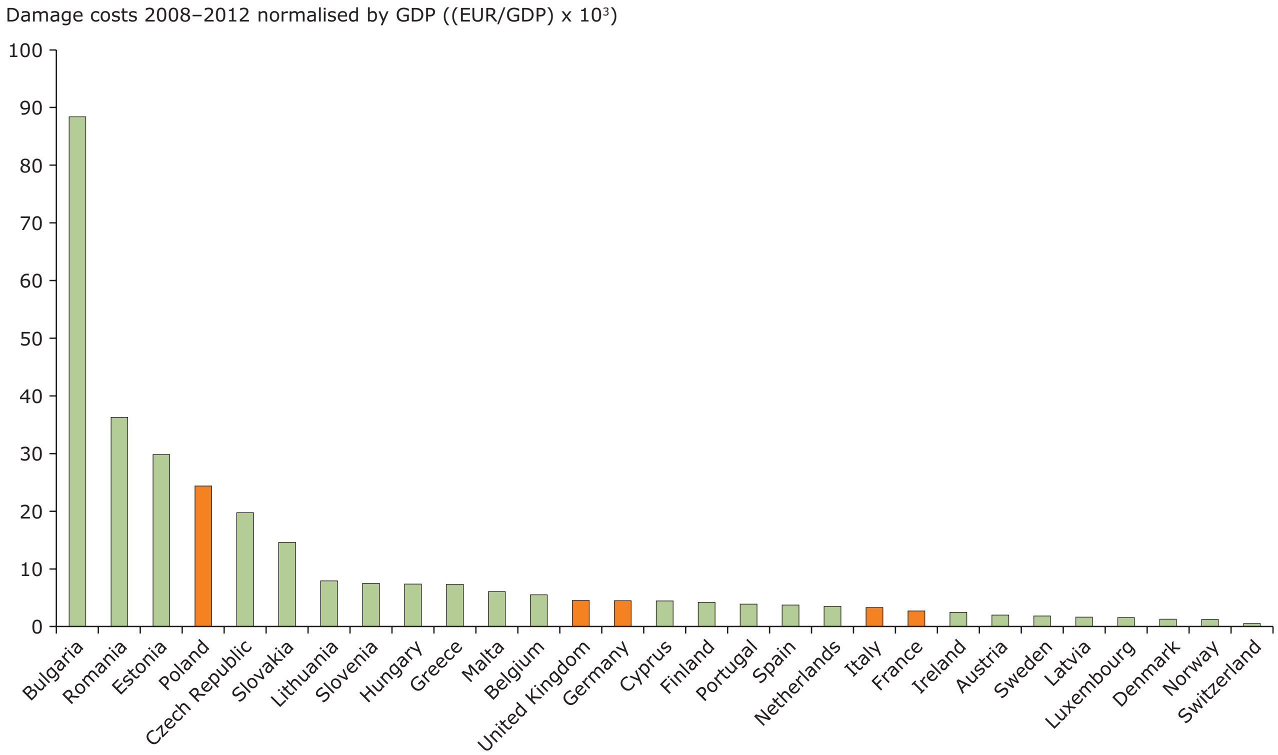
The ranking of individual facilities by absolute damage costs presented earlier in Table 3.4 clearly will change if results are normalised to account for different operating efficiencies. As noted earlier, many of the facilities having the highest damage costs are large power generating facilities. For these plants, it is possible to use CO_2 emissions as a proxy for fuel consumption because CO_2 emissions will have a closer relationship with power production and productivity than any of the other data available for the E-PRTR facilities.

Besides reporting to the E-PRTR, large combustion plants separately report emissions of certain pollutants, and also fuel combustion data, under the EU Large Combustion Plant (LCP) Directive (2001/80/EC) (EU, 2001a). The latest year for which LCP Directive data are publicly available is 2009 (EEA, 2012). A verification of the approach used to normalise damage costs was therefore performed for the top 30 power generating facilities for which data were also available under the LCP Directive reporting, by comparing:

(i) the estimated damage costs from facilities when CO_2 emissions were used as a proxy for the amount of fuel consumed, and

(ii) the estimated damage costs from facilities when actual reported fuel consumption data were instead used as the basis for the normalisation.

Figure 3.10 Aggregate damage costs by country normalised against GDP, 2008–2012

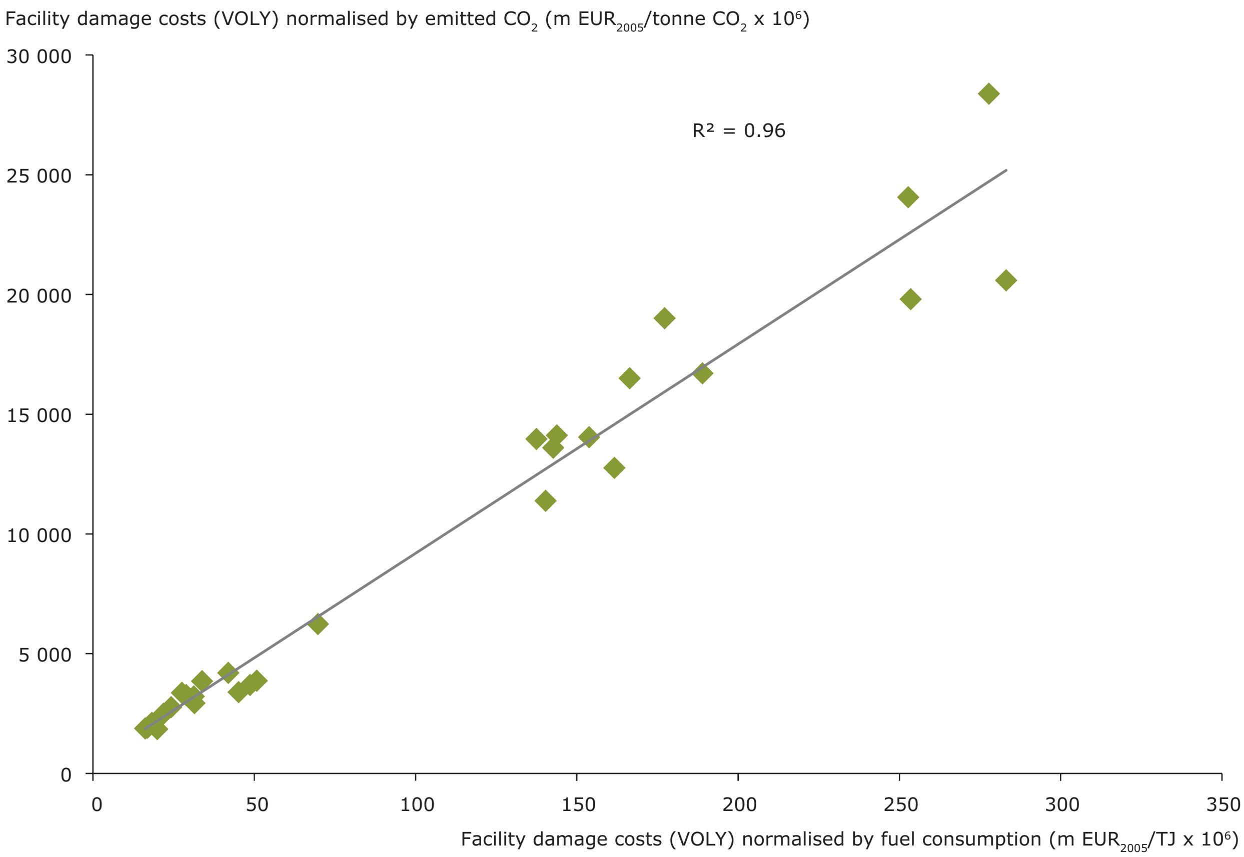


Note: The orange bars highlight the countries with the highest absolute damage costs in Figure 3.8. The ranking is based on the low VOLY approach for mortality valuation for the main air pollutants and a CO_2 price of EUR_{2005} 9.5 per tonne.

Figure 3.11 shows the correlation of the normalised damage costs estimated using the two different normalisation approaches. The high r^2 coefficient indicates the good degree of correlation between the results of ranking using the two different normalisation approaches confirming that normalising by CO_2 emissions can serve as a good approximation of fuel consumption at these facilities. It should be noted, however, that while this correlation is relatively strong for power-generating facilities as shown here, for other types of industrial facilities, the relationship may be weaker.

Unfortunately, the LCP Directive dataset could not be used to a greater extent in this work. There is no official linking or coding system that easily allows facilities in the E-PRTR database to be matched with those included within the LCP Directive reporting. Facilities can sometimes be manually matched (as in this instance), but this is impractical when large numbers of facilities are considered. A better streamlining of industrial data reporting within the EU would facilitate both verifications of the officially reported data, and also increase the usefulness of the respective datasets for assessment purposes.

Figure 3.11 Correlation between normalised damage costs calculated using i) CO_2 emissions as the basis for normalisation and ii) fuel consumption, for the top 30 power generating facilities in 2009



Note: A CO_2 price of EUR_{2005} 9.5 per tonne is applied.

Table 3.5 shows the top 30 ranked facilities when a normalisation of damage cost per unit CO_2 emissions is extended to all the *ca.* 1 200 facilities identified in E-PRTR as power generating facilities and for which CO_2 emissions were reported. One difference when damage costs from individual facilities are normalised by CO_2 emissions is that more large combustion plants from eastern Europe appear in the list of top 30 ranked facilities, suggesting that they contribute more damage cost per unit of fuel consumption, i.e. they are less environmentally efficient.

Many of the facilities that were previously included in the top 30 now appear down the ranking, with changes in their relative ranking also occurring. To illustrate, the top five facilities originally shown in Table 3.4 now appear in positions 25, 248, 22, 320 and 299. However, seven facilities appear in both Table 3.4 and Table 3.5 indicating that their respective estimated damage costs, both in absolute terms and also when normalised, rank among the highest in Europe. These facilities include three in Bulgaria and Romania, and one in Slovakia.

It is also noted that while eight facilities located in Germany were included in the original top 30 list of facilities having the highest absolute damage costs, the top 30 facilities ranked by normalised damage costs contains no German facilities indicating their apparent relative efficiency compared to power generating facilities in other countries.

Table 3.5 Aggregated damage costs for 2008–2012 for the top 30 power generating facilities normalised per unit CO_2 emissions as a proxy for relative operating efficiency

Number	Facility name	City	Country	Aggregated damage costs EUR_{2005} million/t CO_2 x 10^6		Original ranking without normalisation by CO_2
				VOLY low	VSL high	
1	Central de Escucha	Escucha	Spain	342	941	85
2	Miejskie Przedsiębiorstwo Energetyki Cieplnej Sp. z o.o., Ciepłownia Centralna	Chełm	Poland	330	898	1 321
3	TETs 'Maritsa' AD Dimitrovgrad	Dimitrovgrad	Bulgaria	281	865	96
4	Evonik Cofrablack	Ambes	France	256	712	543
5	TPP 'Brikel'	Galabovo	Bulgaria	243	746	28
6	Energomedia Sp. z o.o.	Trzebinia	Poland	243	657	1 610
7	SC Electrocentrale Oradea SA	Oradea	Romania	239	685	42
8	SC CET Govora SA	Ramnicu Valcea	Romania	221	628	35
9	SC CET Arad SA — pe lignit	Arad	Romania	206	586	155
10	Regia Autonoma Pentru Activitati Nucleare — Sucursala Romag Termo	Drobeta Turnu Severin	Romania	196	557	11
11	Wojewódzkie Przedsiębiorstwo Energetyki Cieplnej w Legnicy S. A., Centralna Ciepłownia w Legnicy	Legnica	Poland	182	494	1 805
12	Dunai Gőzfejlesztő Kft.	Százhalombatta	Hungary	180	473	675
13	TETs 'Republika'	Pernik	Bulgaria	177	544	139
14	Przedsiebiorstwo Energetyczne 'Megawat' Sp. z o.o., Zakład Z-2 'Knurów'	Knurów	Poland	177	483	1 281
15	Slovenské elektrárne a.s.-Elektrárne Nováky, závod	Zemianske Kostoľany	Slovakia	151	417	18
16	SC Electrocentrale Deva SA	Mintia	Romania	150	417	17
17	TETS 'Sviloza'	Svishtov	Bulgaria	150	455	146
18	Complexul Energetic Turceni	Turceni	Romania	149	417	3
19	CET Timisoara Sud	Timisoara	Romania	147	412	479
20	SC CET SA Bacau I	Bacau	Romania	143	400	383
21	'TETs Maritsa iztok 2' EAD	Kovachevo	Bulgaria	142	425	1
22	RWE npower plc, Fawley Power Station	Southampton	United Kingdom	141	388	1 465
23	TETs 'Bobov dol'	Golemo selo	Bulgaria	140	421	23
24	Central Diesel Ceuta	Ceuta	Spain	139	345	557
25	Sucursala Electrocentrale Craiova II	Craiova	Romania	137	382	38
26	SC Centrala Electrica de Termoficare Brasov SA	Brasov	Romania	135	376	374
27	Středisko energetiky Důl ČSM — Teplárna Dolu ČSM	Stonava	Czech Republic	129	355	1 895
28	S.C. Uzina Termoelectrica Giurgiu S.A.	Giurgiu	Romania	127	352	1 240
29	Captain FPSO	-	United Kingdom	125	333	2 200
30	Cabot France	Berre-l'Etang	France	122	329	1 132

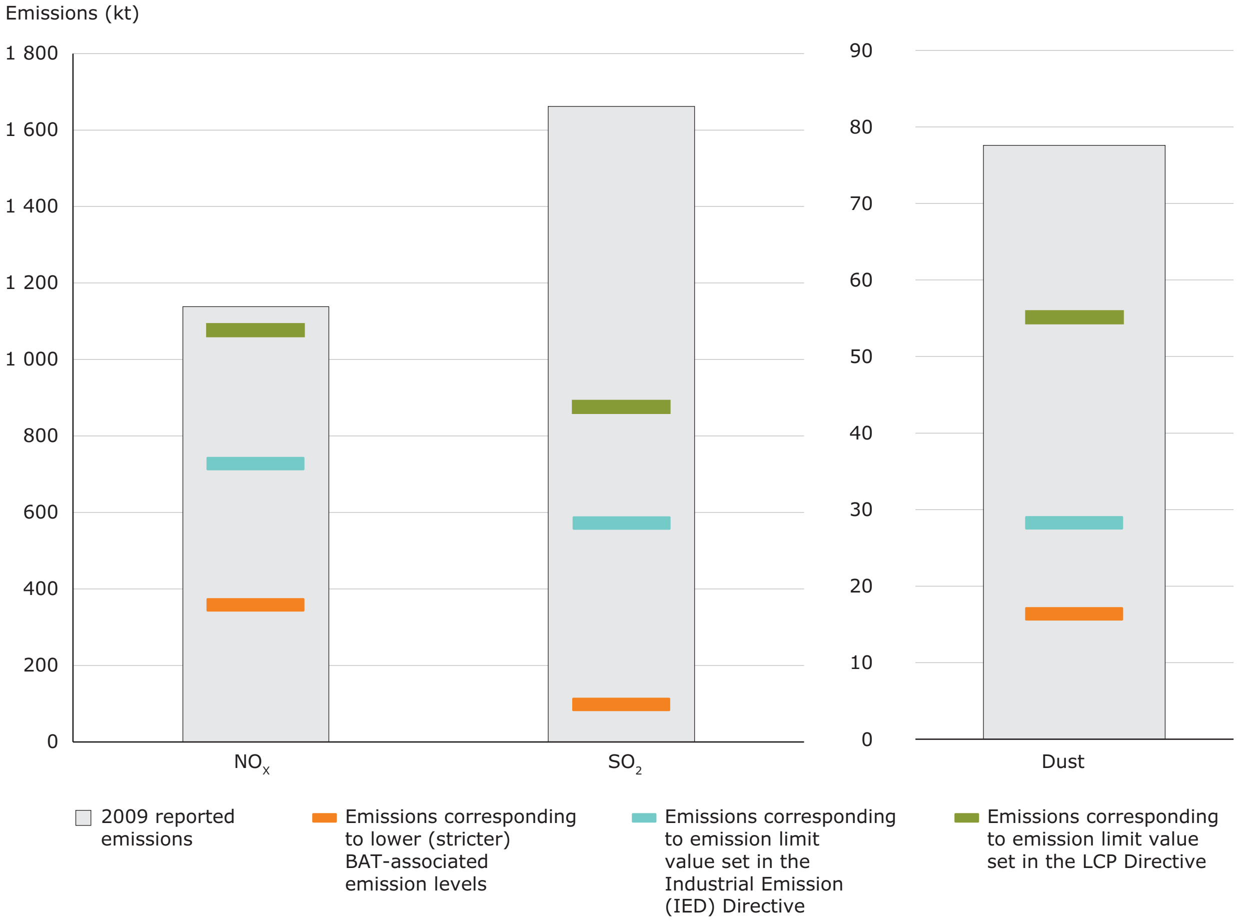
Note: Shaded cells indicate those facilities also included in the original list of top 30 ranked facilities (Table 3.4). A CO_2 price of EUR_{2005} 9.5 per tonne is applied. The order of ranking is based on the lower VOLY approach for the main air pollutants.

3.3 Reducing damage costs of air pollution — a case study for large combustion plants

As an example of the wider application of the methods for estimating air pollution related damage costs described in the preceding sections, this section draws on the results from a second recent EEA assessment. This illustrates the scale of potential benefits in terms of reduced damage costs that would occur were certain industrial facilities to reduce future levels of emissions.

The EEA report *Reducing air pollution from electricity-generating large combustion plants in the European Union* (EEA, 2013b) investigated the hypothetical emission reduction potential of NO_X, SO_2 and dust from more than 1 500 of Europe's large combustion plants (LCP) that operated in 2009. The assessment was based on the latest available emission and fuel-use data from 2009 reported by Member States under the LCP Directive (EU, 2001a). Specific findings of the study included that EU-27 NO_X emissions from LCPs have the potential to be 36 % lower than in 2009 if all plants were to meet the emission limit values (ELVs) set in the Industrial Emission Directive (2010/75/EU) (IED) (EU, 2010), and 69 % lower if plants were to achieve the low (strictest) end of the BAT-associated emission levels (BAT-AELs) described in the 2006 LCP best available techniques reference document (BREF; European Commission, 2006b) (Figure 3.12). For SO_2, the potential emission reductions were 66 % and 94 %, respectively. For dust, the potential emission reductions were a respective 64 % and 79 %. These reductions were based on improved abatement measures and additional measures such as fuel-switching. Wider energy-system transitions were not accounted for in the study.

Figure 3.12 Reported 2009 EU-27 LCP emissions compared with the future emission limit values of the IED, the existing LCP Directive emission limit values and the LCP BREF lower associated emission levels



Source: EEA, 2013b.

The health and environmental benefits that would arise from these hypothetical country-specific emission reductions for NO_X and SO_2 can be evaluated through application of the damage cost methodologies described in this report. Savings arising from the potential reduction of dust emissions are not available as country-specific damage costs estimations. Results of this evaluation are shown in Table 3.6.

The application of the damage costs methodology described in this report, coupled with the potential emission reduction documented in EEA (2013b), shows that direct benefits savings in the EU-27 would be in the order of EUR_{2005} 11.1 to 32.7 billion per year if the 1 500 LCPs addressed in the report were hypothetically to meet the emission limit values set for NO_X and SO_2 in the Industrial Emission Directive. In reality, emission savings would be greater than the values indicated. It has already been noted that savings arising from the reduction of dust (particulate matter) emissions are not quantified. Similarly, improved abatement techniques may also significantly reduce emissions of other pollutants, including for example NMVOCs, heavy metals and organic pollutants. As the size of the potential emission reductions for these pollutants were not evaluated in the original EEA study, the benefits that would occur from a future reduction of these pollutants cannot be estimated here. It is nevertheless clear that regardless of the choice of damage cost values and methodologies employed, substantial future health and environmental benefits would result if emissions of pollutants are reduced.

Table 3.6 Country-specific savings (EUR_{2005} million) arising from the hypothetical emission reduction potential of NO_X and SO_2 from more than 1 500 of Europe's large combustion plants that operated in 2009

Member State	Lower BAT AEL		LCP ELV		IED ELV	
	VOLY low	VSL high	VOLY low	VSL high	VOLY low	VSL high
Austria	33	96	4	12	6	17
Belgium	70	206	8	24	21	63
Bulgaria	2 092	6 526	1 520	4 790	1 847	5 787
Cyprus	15	32	7	14	13	28
Czech Republic	1 403	4 041	159	464	569	1 645
Denmark	24	70	2	6	7	21
Estonia	245	698	149	426	199	568
Finland	91	252	7	18	42	116
France	290	818	104	296	183	517
Germany	2 385	7 101	11	34	69	203
Greece	823	2 343	497	1 446	656	1 877
Hungary	155	448	3	7	17	48
Ireland	202	583	98	285	153	443
Italy	927	2 861	99	303	459	1 418
Latvia	2	6	1	3	2	4
Lithuania	48	139	32	93	42	121
Luxembourg	1	4	0	0	0	0
Malta	9	27	8	23	8	23
Netherlands	133	396	0	0	8	25
Poland	3 370	9 491	612	1 743	2 076	5 846
Portugal	64	169	3	7	25	60
Romania	3 769	11 033	2 770	8 161	3 300	9 687
Slovakia	42	118	2	7	19	55
Slovenia	150	443	0	0	18	51
Spain	608	1 637	198	543	351	941
Sweden	10	28	0	1	4	13
United Kingdom	2 028	5 830	366	1 055	1 082	3 099
EU-27	**18 991**	**55 394**	**6 662**	**19 764**	**11 176**	**32 676**

4 Discussion

The preceding chapters described the development and application of an updated methodology to determine the human health and the environmental damage costs arising from the emissions to air reported by industrial facilities to the E-PRTR. Various issues were identified that introduce potential uncertainties into the results, which could affect the robustness of the analyses. The previous EEA report (EEA, 2011) presented a discussion on a number of these aspects, including commentary on the suitability of the methodologies employed and areas in which the analysis could be improved. These issues are still relevant, but rather than repeat the same discussion in this updated report, readers are referred to the earlier publication for further details.

The focus of discussion in the present chapter is upon two key elements:

- ways in which the E-PRTR might be improved for this type of assessment;
- interpretation of the results from this report.

4.1 Improving the E-PRTR and its implementation to facilitate assessments

As highlighted in preceding sections, there are some ways to improve the E-PRTR and its current implementation by countries to facilitate its use for assessment purposes. The following are considered to be the most important.

- **More complete reporting of emissions from individual facilities**. Review of the facilities with the highest estimated damage costs reveals a number of instances of incomplete reporting, including for PM_{10}, as well as heavy metals and organic pollutants. Such omissions clearly bias any ranking of facilities by under-estimating their respective damage costs.
- **Providing information on the fuel consumption or productive output of individual facilities**. This would enable the environmental efficiency of facilities to be compared, in terms of calculating implied emission factors (see e.g. AMEC, 2012 in relation to emissions from large combustion plants) as well as the estimated damage costs per unit of production or fuel consumption. At present, such information is not reported to the E-PRTR so this type of analysis cannot be done. This reduces the value of the analysis to regulators, for example, since they cannot assess the merits of regulating a few large facilities over a larger number of smaller facilities. It also limits the usefulness of the register for members of the public, as a lack of information on facility capacity or production limits the potential for fair comparisons. Some information on fuel combustion at certain large combustion plants is, however, publicly available at the European level under the LCP Directive reporting for most, but not all, Member States. However, as earlier noted, linking E-PRTR information with that reported under the LCP Directive is difficult.
- **More extended data checking at national level**. Recognising the need to improve the quality of data reported to the E-PRTR, the EEA has implemented an annual data review process in recent years, providing feedback to the competent authorities in each country responsible for compiling facility data (e.g. ETC/ACC, 2011). Nevertheless, it is considered that consideration be given to further checking by countries before data are reported to the E-PRTR, particularly to address completeness of data and to identify outlying values. Such checking is to some extent facilitated by the annual updating of the E-PRTR, which allows the identification of facilities whose emissions vary significantly between years.
- **Improved traceability of facilities**. It proved difficult to compare the results calculated for the present study with those from previous works (Holland, 2006; Barrett and Holland, 2008) on a facility-by-facility basis. Part of the problem relates to differences in the annual E-PRTR datasets received by the EEA, in which facilities may change ownership, name and/or national facility identification code. In addition, locational references can also change over time, from a village location to the nearest town or district for example. Similarly, linking E-PRTR data with information reported under other EU legislation such as the Large Combustion Plant Directive is difficult, due to differences in

facility definitions, facility names and identifiers etc. It is noted in this context that the European Commission is presently undertaking work with a view to ensure the future linking of large combustion plants with E-PRTR, as well as for streamlining of reporting between the IED and E-PRTR.

- **More information on nature of installations**. The sectoral analysis shown in Chapter 3 suffers as a result of the fairly limited sectoral disaggregation available. It would perhaps be helpful if there was a more detailed breakdown of installation types and combustion capacities so that instead of the energy sector, one could, for example, distinguish large power stations from smaller combustion plant that are merely part of a manufacturing facility. Again, the potential future linking of E-PRTR with the reporting made under the LCP Directive would be beneficial in this respect.

While these suggestions are put forward for potentially improving the E-PRTR, the register is nevertheless still recognised as being an extremely useful resource for researchers and members of the public interested in the transparency of environmental information.

4.2 Interpreting the results of this study

The E-PRTR already provides substantial useful information for a variety of users. For example, emissions data show how the major polluters in Europe contribute to the overall pollution burden, and changes in emissions from these facilities could provide an indication of the effectiveness of legislation to reduce the pollutant burdens imposed on society by industry. It is important to note that neither E-PRTR nor this report in any way assess whether the emissions of a facility are consistent with its legal requirements for operating.

The main insight provided by this report is the expression of industrial pollution problems in terms of the impacts and damage costs caused. The knowledge that a given quantity of pollution released to air from a particular location will cause a quantifiable increase in mortality and various kinds of morbidity (e.g. new incidence of chronic bronchitis, restrictions to normal activity, use of medication), along with the associated costs, helps convey the real nature of pollution problems in a way that a simple measure of emissions cannot.

Quantifying effects in monetary terms provides information relevant to the cost-benefit analysis of pollution controls. Information regarding the size of pollution damage can easily be coupled with approximate estimates of the costs of abatement for a preliminary cost-benefit analysis (see Barrett and Holland, 2008). In this context, it is important that the benefits of industrial facilities (such as producing goods and products, and generating employment and tax revenues) are properly recognised, and not just the costs. These benefits are not addressed in this report.

Concerning the monetisation of damage costs, it has already been noted that, in the absence of any accepted unified methodology, different methods are used in this assessment to estimate the damage cost per tonne emitted for the different pollutants i.e. the main air pollutants, CO_2, heavy metals and organic pollutants. For the main air pollutants and CO_2, a range of selected values was used to provide an indication of the different methods and the uncertainty inherent in the different approaches presently used to value emissions for policy appraisal purposes. Thus, while the values used to estimate damage costs will inevitably change in the future as new knowledge develops, it would be unlikely to alter the conclusion that the damage costs associated with air pollution and CO_2 emissions from E-PRTR facilities are likely to be very significant. The basis of the methods used for determining damage costs for the main air pollutants have been developed over many years and have been extensively reviewed at the European level — they are therefore considered reasonable mature. As recommended in Chapter 2, it is clear, however, that a wider debate is still required on how better to estimate the economic impacts of changes in greenhouse gas emissions.

In summary, this report has presented an updated methodology that allows for the estimation of damage costs caused by emissions of selected pollutants from industrial facilities included in the E-PRTR. It demonstrates that, compared to using emissions data alone, these methods provide additional insights into the costs of harm caused by industrial air pollution. Such insights are particularly valuable in the context of current discussions in Europe on how best to move towards a resource-efficient and low-carbon economy. Moreover, the analysis might be further strengthened in the future by integrating efficiency and productivity data for individual facilities into the analysis of damage costs.

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Annex 1 Corrections made to reported E-PRTR emissions data

Table A1.1 shows instances where E-PRTR data points were revised to correct apparent errors in the reported emissions or units. These anomalies were clearly identified either because there was one value amongst a time-series of emission values which differed by an order of magnitude or more, or because the reported value was a significant outlier while other pollutants reported from the same facility were consistent with the magnitude of emissions reported from other facilities.

Table A1.1 Revised emission values used in the assessment following quality checks performed on the E-PRTR data set

Facility ID	Facility name	City	Country	Activity code	Activity name	Year	Pollutant	Unit	Reported emissions	Revised emissions
5951	Eesti Energia Narva Elektrijaamad AS, Balti elektrijaam	Narva linn	Estonia	1.(c)	Thermal power stations and other combustion installations	2011	PM_{10}	kg	20800000	10800000
5952	Eesti Energia Narva Elektrijaamad AS, Eesti soojuselektrijaam	Auvere küla, Vaivara vald	Estonia	1.(c)	Thermal power stations and other combustion installations	2008	As and compounds	kg	7240	724
5952	Eesti Energia Narva Elektrijaamad AS, Eesti soojuselektrijaam	Auvere küla, Vaivara vald	Estonia	1.(c)	Thermal power stations and other combustion installations	2009	As and compounds	kg	6120	612
5952	Eesti Energia Narva Elektrijaamad AS, Eesti elektrijaam	Auvere küla, Vaivara vald	Estonia	1.(c)	Thermal power stations and other combustion installations	2010	As and compounds	kg	8100	810
5952	Eesti Energia Narva Elektrijaamad AS, Eesti elektrijaam	Auvere küla, Vaivara vald	Estonia	1.(c)	Thermal power stations and other combustion installations	2011	As and compounds	kg	7320	732
5952	Eesti Energia Narva Elektrijaamad AS, Eesti elektrijaam	Auvere küla, Vaivara vald	Estonia	1.(c)	Thermal power stations and other combustion installations	2012	As and compounds	kg	8000	800
4365	Semariv- CITD	Vert-Le-Grand	France	5.(b)	Installations for the incineration of non-hazardous waste	2011	NO_X	kg	15800000	158000
18023	Eurocopter Marignane	Marignane	France	2.(f)	Installations for surface treatment of metals and plastic materials	2012	Ni and compounds	kg	203000	203
104967	SWN Stadtwerke Neumünster GmbH	Neumünster	Germany	1.(c)	Thermal power stations and other combustion installations	2008	CO_2	kg	3.98E+11	398000000
104967	SWN Stadtwerke Neumünster GmbH	Neumünster	Germany	1.(c)	Thermal power stations and other combustion installations	2009	CO_2	kg	4.03E+11	403000000

Table A1.1 Revised emission values used in the assessment following quality checks performed on the E-PRTR data set (cont.)

Facility ID	Facility name	City	Country	Activity code	Activity name	Year	Pollutant	Unit	Reported emissions	Revised emissions
104967	SWN Stadtwerke Neumünster GmbH	Neumünster	Germany	5.(b)	Installations for the incineration of non-hazardous waste	2010	CO_2	kg	4.18E+11	418000000
104967	SWN Stadtwerke Neumünster GmbH	Neumünster	Germany	1.(c)	Thermal power stations and other combustion installations	2011	CO_2	kg	4.26E+11	426000000
46335	Hawle Guss GmbH	Fürsten-walde/ Spree	Germany	2.(d)	Ferrous metal foundries	2012	PCDD+PCDF (dioxins + furans)	kg	5.34	0.00534
191906	'Eko Osta' SIA	Rīga	Latvia	5.(a)	Installations for the recovery or disposal of hazardous waste	2012	PCDD+PCDF (dioxins + furans)	kg	2	0.002
2	AB 'Achema'	Jonalaukis	Lithuania	4.(c)	Chemical installations for the production on an industrial scale of phosphorous-, nitrogen- or potassium-based fertilisers	2011	CO_2	kg	112000000	1120000000
2	AB 'Achema'	Jonalaukis	Lithuania	4.(c)	Chemical installations for the production on an industrial scale of phosphorous-, nitrogen- or potassium-based fertilisers	2012	CO_2	kg	212000000	2120000000
14409	Delimara Power Station	Marsaxlokk	Malta	1.(c)	Thermal power stations and other combustion installations	2011	Ni and compounds	kg	149	1490
9188	DMercedes-Benz España, S.A.	Vitoria-Gasteiz	Spain	9.(c)	Installations for the surface treatment of substances, objects or products using organic solvents,	2010	NMVOC	kg	10000000000	1000000
9188	DMercedes-Benz España, S.A.	Vitoria-Gasteiz	Spain	9.(c)	Installations for the surface treatment of substances, objects or products using organic solvents	2010	PM_{10}	kg	11100000000	111000
32810	Murco Petroleum Limited, Milford Haven Refinery	Pembroke-shire	United Kingdom	1.(c)	Thermal power stations and other combustion installations	2011	NH_3	kg	338000	33800
32810	Murco Petroleum Limited, Milford Haven Refinery	Pembroke-shire	United Kingdom	1.(c)	Thermal power stations and other combustion installations	2012	NH_3	kg	307000	30700

Annex 2 Determination of country-specific damage cost per tonne estimates for the main air pollutants

A2.1 Overview

This annex addresses the methods for quantifying damage costs for the major main air pollutants: NH_3, NO_X, PM_x, SO_2 and NMVOCs. Analysis follows the impact pathway methodology originally developed in the ExternE Project funded by the European Commission's DG Research (ETSU/Metroeconomica, 1995; Holland et al., 1999; Bickel and Friedrich, 2005) with further refinement from the CAFE Programme (Holland et al., 2005a and 2005b; Hurley et al., 2005) and recent work in the context of revision of the European Commission's Thematic Strategy on Air Pollution (WHO, 2013a, b; Holland, 2014a, b).

The dispersion modelling tracks pollutants through the atmosphere and follows their chemical reactions, enabling quantification of effects linked to emissions, not simply to the atmospheric concentration of the pollutant in the chemical state in which it was released. An important consequence is that effects caused by secondary particulates are not assigned to $PM_{2.5}$ but to the primary pollutant from which they are formed (e.g. SO_2 for sulphate aerosol, NO_X for nitrate aerosol and NH_3 for ammonium aerosol). It also enables accounting for less obvious interactions between air pollutants, for example the effects of NMVOC emissions on inorganic particle concentrations, or the effects of NO_2 and NH_3 emissions on ground-level (tropospheric) ozone formation.

The price year used is 2005, for consistency with, for example, the cost benefit analysis recently performed by the European Commission in support of the proposed Clean Air Policy Package (European Commission, 2013a; 2013c).

A2.2 Impacts considered and omitted from the analysis

The impacts that have been quantified for this report are listed in Table A2.1. It is important not to forget those effects that remain unquantified as a result of limitations in the availability of data on response functions and/or valuation. These are listed in Table A2.2, which shows that a large number of effects have not been quantified.

To interpret the information presented in the two tables, it is important to be aware that:

1. the effects that have been quantified are substantial;
2. several of the effects that have not been quantified here are likely to be negligible (e.g. direct effects of SO_2 and NO_X on crops) and would not lead to a significant increase in damage per tonne of emissions;
3. the value of certain ecosystem effects (not quantified in this report) may also be substantial.

In summary, while omitting any impact leads to a bias to underestimate damages and some of the omitted effects are undeniably important, the results generated here quantify a large fraction of total damages for most of the pollutants considered.

The effect of omitting impacts should be seen in the context of the full range of uncertainties in the assessment. While it clearly biases towards underestimation, the full set of uncertainties, including also model assumptions and statistical uncertainties, may push the results either up or down. More information on these uncertainties is provided in the third volume of the CAFE CBA methodology (Holland et al., 2005c).

It is necessary to consider the recommendations of the WHO HRAPIE study concerning the quantification of chronic effects of NO_2 on mortality, to be performed in areas where NO_2 levels are above 20 ug/m^3. Inclusion of these effects would have a significant impact on NO_2 related impacts e.g. for the United Kingdom it is estimated that a factor 20 difference would occur between unit change in exposure above and below the threshold. The resolution of the modelling undertaken in this assessment is too coarse to accurately identify areas where the threshold is exceeded and the size of the

Table A2.1 Quantified impacts for the main air pollutants

Burden	Effect
Human exposure to $PM_{2.5}$	Chronic effects on: Mortality Adults over 30 years Infants Morbidity Bronchitis in adults Bronchitis in children Acute effects on: Morbidity Respiratory hospital admissions Cardiac hospital admissions Consultations with primary care physicians Restricted activity days Work loss days Asthma symptoms in children
Human exposure to ozone	Acute effects on: Mortality Morbidity Respiratory hospital admissions Cardiac hospital admissions Minor restricted activity days
Human exposure to NO_2	Chronic effects on: Morbidity: Bronchitis in asthmatic children Acute effects on: Mortality Morbidity: Respiratory hospital admissions
Exposure of crops to ozone	Yield loss for: barley, cotton, fruit, grape, hops, millet, maize, oats, olive, potato, pulses, rapeseed, rice, rye, seed cotton, soybean, sugar beet, sunflower seed, tobacco, wheat
SO_2 effects on utilitarian buildings	Degradation of: stone and metalwork, particularly zinc, galvanised steel

Table A2.2 Effects omitted from the analysis of main air pollutants

Effect	Comments
Health	
Ozone chronic — mortality chronic — morbidity	Function for chronic impacts on mortality identified by WHO (2013b), but not recommended for core analysis
NO_2 chronic — mortality	Function for chronic impacts on mortality identified by WHO (2013b), for application only above a threshold of 20 ug/m^3
Direct effects of SO_2, NMVOCs	
Social impacts	Limited data availability
Altruistic effects	Reliable valuation data unavailable
Agricultural production	
Direct effects of SO_2 and NO_x	Negligible according to past work
N deposition as crop fertiliser	Negligible according to past work
Visible damage to marketed produce	Locally important for some crops
Interactions between pollutants, with pests and pathogens, climate etc.	Exposure-response data unavailable
Acidification/liming	Negligible according to past work
Materials	
Effects on cultural assets, steel in reinforced concrete	Lack of information on the asset stocks at risk and valuation data
PM and building soiling	
Effects of O_3 on paint, rubber	
Ecosystems	
Effects on biodiversity, forest production, etc. from excess O_3 exposure, acidification and nitrogen deposition	Valuation of ecological impacts is currently considered too uncertain
Visibility	
Change in visual range	Impact of little concern in Europe
Drinking water supply and quality	Limited data availability

population subject to exceedance, and so chronic effects of mortality are necessarily excluded from the analysis. The extent to which this may affect the analysis requires consideration. Road traffic is generally the dominant source of NO_2 in most locations where exceedances of the threshold occur. Inevitably, even in these locations, there will be some background contribution from industrial sources and it may be argued that in areas where there is exceedance the contribution of industrial facilities should be valued at the marginal rate (including the chronic mortality impacts). An alternative view is that, for locations where there are exceedances of the threshold, it would be more logical to target policy on the dominant (local) sources (in most cases, traffic). For this analysis this seems a more realistic position, and so quantification of NO_2 impacts using only the functions dealing with acute exposure is considered to be a better approach than would be given from additional inclusion of chronic effects on mortality where the threshold is exceeded. It should be noted that this discussion is specific only to damage associated with NO_2 exposure, the assessment of exposure to secondary nitrate particulate matter also derived from NO emissions, includes quantification of effects of chronic exposure on mortality.

A2.3 Other uncertainties considered

In addition to the uncertainty arising from omitting a number of impacts from the analysis, the earlier analysis by Holland et al. (2005c) specifically addressed some other key uncertainties and sensitivities:

- valuation of mortality using the value of statistical life (VSL) and value of a life year (VOLY) approaches;
- quantifying ozone effects on health with and without a 'cut-point' (effectively, the assumption of a threshold at 35 ppb);
- separating health impacts into a 'core' set of functions that are determined to be most robust and a 'sensitivity' set of functions that are less robust.

A conclusion drawn from the earlier work was that the uncertainty in mortality valuation was dominant, and so this is the main quantified uncertainty carried into the present study.

An important issue that has not been addressed relates to uncertainty in apportioning impacts to each pollutant. This is most problematic for quantifying the impacts of fine PM, which are typically described by epidemiological studies in terms of PM_{10} or $PM_{2.5}$ rather than the constituent species of PM (e.g. sulphate aerosol, combustion particulate matter, natural material). The review of health aspects of air pollution in Europe performed by WHO (2004), did not attempt to differentiate between PM. This position has been retained for the updated work by WHO in the REVIHAAP and HRAPIE studies (WHO, 2013a, b).

A2.4 Development of source-receptor relationships

Source-receptor relationships define the link between the site of emission and the site of impact. These have been developed using data provided from the EMEP chemical transport model to the International Institute for Applied Systems Analysis (IIASA) for the revision of the Thematic Strategy on Air Pollution in 2013 (Heyes, Oxley, personal communications).

These data cover a variety of pollutants, the primary species emitted and their reaction products. For each EMEP model run the analysis adjusts by 15 % the emissions of one pollutant in one country for one baseline year. This is repeated until all combinations of pollutants, countries and baseline year have been modelled. For the purpose of the present analysis, the change in pollutant concentration or deposition is then divided by the quantity of pollutant adjusted in each model run, to derive a change per tonne of emission. The emission scenario year used was 2010.

'Source-receptor (SR) matrices give the change in various pollution levels in each receptor country (or grid square) resulting from a change in anthropogenic emissions from each individual emitter. Such matrices are generated by reducing emissions for each emitter of one or more precursors by a given percentage (15 % in this case), running the EMEP model with these reduced emissions, and comparing the resulting output fields with the base simulation, i.e. a simulation without any emission reduction. The reason for this procedure is to keep the chemical conditions as close to the original conditions as possible.'

Source: EMEP, 2005.

The EMEP modelling includes not only dispersion of the primary pollutants i.e. pollutants in the chemical form in which they were released to the atmosphere, but also the subsequent chemical reactions of these pollutants in the atmosphere. Hence it accounts for the formation of ozone (linked to emissions of NO and NMVOCs), secondary inorganic aerosols (ammonium sulphate and ammonium nitrate from release of NH_3, NOX and SO_2) and secondary organic aerosols (from release of certain NMVOCs). The inclusion of secondary organic aerosols is new to this round of modelling. The EMEP model generated transfer matrices for 5 meteorological years (2006 to 2010) from which averages were taken, recognising that meteorological variability has a significant impact on pollutant chemistry and dispersion.

The steps taken to process the EMEP outputs are outlined below.

1. Each 15 % reduction file was subtracted from the baseline to provide the difference in concentration per grid cell by substance, reduced pollutant and emitting country.

2. The concentration in each grid cell was multiplied by the population (population by grid cells taken from EMEP data) in that grid cell to generate a population-weighted average change in concentration.

3. The change in concentration in each grid cell was divided by the total annual emissions for each country to generate the change in concentration per tonne emission of each of the five emitted pollutants (SO_2, NO_X, NMVOCs, NH_3 and $PM_{2.5}$). The total annual emissions were provided by EMEP.

4. The population-weighted values were multiplied by the population at risk (total population, over 65s, children, etc.), health concentration-response functions and the values associated with each type of health impact.

5. These country-specific damage costs were then multiplied by the E-PRTR facility emissions data to provide the estimated damage costs from each E-PRTR facility.

The updated results in terms of damage cost per tonne of pollutant emission are different to those applied in the earlier report (EEA, 2011) for a number of reasons, including:

- updated dispersion modelling from the EMEP model including certain technical improvements and corrections since implemented in the model;

- the use of a 2010 emissions baseline scenario coupled with a 5-year average meteorology;

- adoption of recommendations from the recent World Health Organization's (WHO) HRAPIE study (WHO, 2013a); and

- refinement of certain pollutant-specific effects e.g. for chronic bronchitis.

When generalising results of dispersion modelling, there may be problems from non-linearity of some of the atmospheric processes, most notably those dealing with ozone and hence linked to emissions of NO_X and NMVOCs. However, these are not considered too problematic here for several reasons. Most importantly, ozone effects generate only a small amount of the overall pollution damage, with effects of fine PM being far more significant. Recent analysis for the Gothenburg Protocol suggests that over 95 % of health damage from the main air pollutants is attributable to PM. It may be argued that the role of ozone is being underestimated, perhaps through the omission of some types of effect, but ozone-related damage would need to increase very markedly for this to be a problem. The WHO-Europe HRAPIE and REVIHAAP studies (WHO 2013a, 2013b) highlight chronic effects of exposure to ozone on mortality, though only for sensitivity analysis.

A2.5 Quantification of health damage

Full details of the response functions, incidence data and valuations are given in Holland (2014a, b). Response functions and valuations are listed here in Table A2.3 for effects of exposure to $PM_{2.5}$, Table A2.4 for effects of exposure to ozone and Table A2.5 for effects of exposure to NO_2. Median VOLY and mean VSL values are used. EEA (2011) provided additional columns in the equivalent tables showing incidence rates, relevant fractions of population, etc. The revised analysis here uses country specific data (see Holland, 2014a).

It should be noted that:

- chronic mortality estimates for $PM_{2.5}$ based on VSL/VOLY or median/mean estimates are not additive but are used as alternatives in sensitivity analysis;

- similarly, for the VOLY mean and median valuations listed for ozone;
- valuation of ozone mortality impacts using the VOLY approach assumes an average loss of life expectancy amongst those affected of one year;
- valuation data refer to the year 2005.

For ozone, effects are quantified against the metric SOMO35 for European analysis (sum of mean ozone over 35 parts per billion).

A2.6 Quantification of ozone crop damage

The analysis of crop damage included here has not been updated since EEA (2011). It is based on the use of AOT40 relationships, combined with EMEP estimates of change in AOT40 on a 50 x 50 km grid. The functions and pollution data have been adjusted, as outlined below.

- The AOT40 outputs from EMEP are for the period May–July. These have been adjusted by country-specific factors derived from earlier EMEP model runs to better represent the growing season for each country.
- The EMEP data are generated for a height of three metres. This has been adjusted to canopy height for each crop based on default relationships in the ICP Mapping and Modelling Manual (ICP Modelling and Mapping, 2004).

Functions and other data are shown in Table A2.5. Valuation data are based on world market prices reported by the United Nations Food and Agriculture Organization. The height factor accounts for variation in ozone concentration with height and is based on default estimates in ICP Mapping and Modelling Manual (2004).

More recent estimates of crop damage from ozone have been derived using flux-based response functions (e.g. Mills and Harmens, 2011) rather than the concentration based functions used here. Analysis during the work for the European Commission on the review of the Thematic Strategy on Air Pollution considered the consistency of these different estimates and found reasonable agreement for the change in damage under different scenarios.

Table A2.3 Incidence data, response functions and valuation data for quantification of health damages linked to PM exposure for 2010 (2005 prices)

Effect	Relative risk per 10 µg.m^{-3}	Valuation (EUR)
Chronic mortality (deaths, VSL valuation)	1.062	2 200 000
Chronic mortality (life years lost, VOLY valuation)	1.062	57 700
Infant mortality (1–12 months)	1.04	3 300 000
Chronic bronchitis, population aged over 27 years	1.117	53 600
Chronic bronchitis, children aged 6–12 years	1.08	588
Respiratory hospital admissions, all ages	1.019	2 220
Cardiac hospital admissions, all ages	1.0091	2 220
Restricted activity days (RADs) working age population	1.047	92
Work days lost	1.046	130
Incidence of asthma symptoms in asthmatic children aged 5–19 years	1.028	42

Note: Response function expressed as relative risk per 10 µg.m^{-3} $PM_{2.5}$.

Table A2.4 Incidence data, response functions and valuation data for quantification of health damages linked to ozone exposure for 2010 (2005 prices)

Effect	Relative risk per 10 µg.m^{-3}	Valuation (EUR)
Acute mortality (life years lost, VOLY median valuation)	1.0029	57 700
Acute mortality (life years lost, VOLY mean valuation)	1.0029	133 000
Respiratory hospital admissions, ages over 65	1.0044	2 220
Cardiac hospital admissions, ages over 65	1.0089	2 220
Minor restricted activity days, ages 18–64	1.0154	42

Note: Response function units: impact per 10 µg.m^{-3} 8-hour daily average ozone.

Table A2.5 Incidence data, response functions and valuation data for quantification of health damages linked to NO_2 exposure for 2010 (2005 prices)

Effect	Relative risk per 10 µg.m^{-3}	Valuation (EUR)
Acute mortality (life years lost, VOLY median valuation)	1.0027	57 700
Acute mortality (life years lost, VOLY mean valuation)	1.0027	133 000
Bronchitic symptoms in asthmatic children aged 5–14 years	1.021	588
Respiratory hospital admissions, all ages	1.018	2 220

Note: Response function expressed as change in incidence rate per 10 µg.m^{-3} NO_2.

Table A2.6 Functions and associated factors for quantification of ozone damage to crop production

Crop	Value (EUR) per tonne	Function	Height (m)	Height factor
Barley	120	0	1	0.88
Fruit	680	0.001	2	0.93
Grapes	360	0.003	1	0.88
Hops	4 100	0.009	4	0.96
Maize	100	0.004	2	0.93
Millet	90	0.004	1	0.88
Oats	110	0	1	0.88
Olives	530	0	2	0.93
Potatoes	250	0.006	1	0.88
Pulses	320	0.017	1	0.88
Rapeseed	240	0.006	1	0.88
Rice	280	0.004	1	0.88
Rye	80	0	1	0.88
Seed cotton	1 350	0.016	1	0.88
Soybeans	230	0.012	1	0.88
Sugar beets	60	0.006	0.5	0.81
Sunflower seed	240	0.012	2	0.93
Tobacco leaves	4 000	0.005	0.5	0.81
Wheat	120	0.017	1	0.88

Note: The function shows proportional change in yield per ppm.hour.

A2.7 Results

The tables below present the estimated damage of pollution, expressed as euros per tonne of emissions of NH_3, NO_X, $PM_{2.5}$, PM_{10}, SO_2 and NMVOCs, for countries throughout Europe. The emissions scenario year for the EMEP modelling used in this study is 2010.

Emissions of SO_2, NO_X and NMVOCs (and to a lesser extent for PM and NH_3) are expected to decline significantly in future years, for example as a result of both current and future European legislation. A good example concerns current legislation on vehicle emissions, which will not be fully effective until the current vehicle fleet is fully replaced. The future change in the overall pollution load of the atmosphere, and changes in the spatial location of emission sources, will also affect the atmospheric chemical reactions between pollutants and, in turn, the associated air pollution related impacts. The damage cost values per tonne pollutant will change with time and should therefore not be assumed to be constant.

Table A2.7 Damage (EUR) per tonne emission estimates for NH_3 and NO_X (2005 prices)

Country		NH_3		NO_X	
		Low VOLY	High VSL	Low VOLY	High VSL
AL	Albania	4 794	10 768	4 082	8 308
AT	Austria	9 914	29 615	8 681	24 442
BA	Bosnia and Herzegovina	8 651	24 282	5 511	14 031
BE	Belgium	19 223	57 437	4 152	12 227
BG	Bulgaria	10 166	33 489	4 588	12 581
BY	Belarus	7 703	22 479	4 033	10 691
CH	Switzerland	6 422	18 856	11 997	33 635
CY	Cyprus	2 194	4 668	593	1 196
CZ	Czech Republic	19 318	56 460	6 420	17 663
DE	Germany	13 617	41 798	6 817	19 059
DK	Denmark	4 693	13 944	3 092	8 515
EE	Estonia	5 017	14 664	2 159	5 566
ES	Spain	4 345	12 224	2 241	5 183
FI	Finland	2 912	8 408	1 481	3 780
FR	France	6 258	18 149	5 463	13 951
GR	Greece	5 085	15 632	1 390	3 142
HR	Croatia	10 477	31 786	6 802	18 433
HU	Hungary	17 191	51 980	7 502	20 354
IE	Ireland	1 692	5 034	3 736	9 785
IT	Italy	11 221	35 689	7 798	23 029
LT	Lithuania	4 914	14 479	3 778	9 935
LU	Luxembourg	16 125	48 130	6 468	17 974
LV	Latvia	5 195	15 651	3 021	7 851
MD	Moldova	13 517	38 902	5 516	14 667
MK	Former Yugoslav Republic of Macedonia	9 125	24 294	3 449	8 349
MT	Malta	4 893	12 756	736	1 696
NL	Netherlands	12 199	35 859	4 854	14 770
NO	Norway	2 507	7 048	1 675	4 081
PL	Poland	13 435	38 240	5 131	13 840
PT	Portugal	4 018	11 921	1 805	4 367
RO	Romania	11 418	33 832	7 507	20 361
SE	Sweden	4 017	12 152	2 197	5 662
SI	Slovenia	14 343	43 277	9 127	25 992
SK	Slovakia	20 436	57 719	6 729	17 936
UA	Ukraine	16 780	51 145	3 800	10 079
UK	United Kingdom	9 503	27 790	3 558	9 948

Table A2.8 Damage (EUR) per tonne emission estimates for $PM_{2.5}$ and PM_{10} (2005 prices)

Country		$PM_{2.5}$		PM_{10}	
		Low VOLY	High VSL	Low VOLY	High VSL
AL	Albania	26 582	55 439	17 261	36 000
AT	Austria	38 300	113 642	24 870	73 794
BA	Bosnia and Herzegovina	20 720	58 677	13 455	38 102
BE	Belgium	57 327	170 702	37 226	110 845
BG	Bulgaria	24 186	80 806	15 705	52 472
BY	Belarus	20 200	59 335	13 117	38 529
CH	Switzerland	55 427	160 225	35 991	104 042
CY	Cyprus	7 015	14 917	4 555	9 686
CZ	Czech Republic	39 882	115 146	25 897	74 770
DE	Germany	47 310	147 553	30 721	95 814
DK	Denmark	16 074	48 050	10 438	31 201
EE	Estonia	9 418	27 684	6 115	17 976
ES	Spain	26 595	74 455	17 269	48 347
FI	Finland	5 942	17 139	3 858	11 129
FR	France	33 751	96 917	21 917	62 933
GR	Greece	18 669	56 883	12 123	36 937
HR	Croatia	21 353	65 336	13 866	42 426
HU	Hungary	38 433	118 336	24 956	76 841
IE	Ireland	13 461	40 315	8 741	26 178
IT	Italy	48 288	154 289	31 356	100 187
LT	Lithuania	15 979	47 453	10 376	30 813
LU	Luxembourg	36 007	105 895	23 381	68 763
LV	Latvia	12 412	37 736	8 060	24 504
MD	Moldova	29 935	85 455	19 439	55 490
MK	Former Yugoslav Republic of Macedonia	19 978	52 814	12 973	34 295
MT	Malta	5 625	15 338	3 653	9 960
NL	Netherlands	54 535	154 240	35 413	100 156
NO	Norway	5 638	15 846	3 661	10 290
PL	Poland	42 153	117 344	27 372	76 198
PT	Portugal	21 129	62 483	13 720	40 573
RO	Romania	35 666	105 101	23 160	68 247
SE	Sweden	7 644	23 204	4 964	15 067
SI	Slovenia	33 836	101 827	21 971	66 122
SK	Slovakia	32 503	92 299	21 106	59 934
UA	Ukraine	29 670	91 284	19 266	59 275
UK	United Kingdom	38 393	111 766	24 930	72 576

Table A2.9 Damage (EUR) per tonne emission estimates for NMVOCs and SO_2 (2005 prices)

Country		NMVOC (including SOA *)		SO_2	
		Low VOLY	High VSL	Low VOLY	High VSL
AL	Albania	839	2 088	8 822	20 069
AT	Austria	2 248	6 184	19 651	58 494
BA	Bosnia and Herzegovina	1 077	2 840	7 601	21 941
BE	Belgium	2 368	5 750	22 591	66 516
BG	Bulgaria	912	2 554	6 238	19 696
BY	Belarus	844	2 174	11 052	32 206
CH	Switzerland	2 946	7 855	30 800	90 337
CY	Cyprus	105	237	1 052	2 270
CZ	Czech Republic	2 075	5 518	12 483	36 491
DE	Germany	1 891	4 772	18 956	57 524
DK	Denmark	1 156	2 756	11 209	33 200
EE	Estonia	670	1 723	5 826	16 692
ES	Spain	1 074	2 690	7 520	21 120
FI	Finland	599	1 544	4 117	11 867
FR	France	1 616	4 087	15 875	45 909
GR	Greece	911	2 386	4 000	11 671
HR	Croatia	1 542	4 159	10 348	31 348
HU	Hungary	1 751	4 830	11 821	35 479
IE	Ireland	1 046	2 647	11 011	32 378
IT	Italy	3 179	8 968	14 729	46 150
LT	Lithuania	794	2 066	10 106	29 748
LU	Luxembourg	2 355	5 891	18 763	55 912
LV	Latvia	866	2 252	8 770	26 175
MD	Moldova	967	2 627	10 602	30 622
MK	Former Yugoslav Republic of Macedonia	990	2 587	6 197	16 862
MT	Malta	674	1 651	2 302	6 895
NL	Netherlands	2 364	5 722	25 269	74 414
NO	Norway	478	1 145	3 878	11 168
PL	Poland	1 610	4 194	11 802	33 613
PT	Portugal	628	1 534	5 216	14 949
RO	Romania	1 159	3 148	10 668	31 439
SE	Sweden	797	2 038	5 209	15 438
SI	Slovenia	2 809	7 882	15 774	47 749
SK	Slovakia	1 442	3 838	10 411	30 093
UA	Ukraine	1 069	2 859	7 029	20 832
UK	United Kingdom	1 450	3 468	14 425	41 861

Note: * SOA — secondary organic aerosols.

Annex 3 Determination of country-specific damage cost per tonne estimates for heavy metals and organic micro-pollutants

A3.1 Objective

The RiskPoll model was used to predict the health impacts and damage costs due to air emissions of the heavy metals arsenic, cadmium, chromium, lead, mercury and nickel and the organic compounds 1,3 butadiene, benzene, diesel particulates, dioxins/furans, formaldehyde and polycyclic aromatic hydrocarbon.

There are alternatives to using RiskPoll, for example the approach and results of the ESPREME project ([9]). Further debate on the differences in methodology between estimates for heavy metal damages is to be welcomed as the models have not been subjected to the same degree of scrutiny as the analysis of the regional pollutants. The modelling of exposure to metals is far more complex, however, requiring a focus on ingestion (in particular), as well as inhalation. Further issues can also arise, for example the probability of surviving cancers caused by different pollutants.

A3.2 Atmospheric dispersion

Air concentrations are calculated using the Uniform World Model (UWM) methodology, described in 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. Continental estimates of deposition velocities for Europe are 0.34 cm/s for arsenic and lead, and 0.57 cm/s for all the other pollutants, except mercury. Country-specific deposition velocities can vary a lot about mean regional estimates. In Europe, for example, the deposition velocity for arsenic ranges from 0.26 to 0.54 cm/s, while for dioxins/furans, the range is 0.43–0.89 cm/s. In both cases, the coefficient of variation is approximately 20 %.

Table A3.1 Country-specific depletion velocities (cm/s) for arsenic and lead (for all other pollutants multiply by 5/3)

Austria	0.35
Balkans	0.29
Belgium	0.40
Bulgaria	0.29
Cyprus	0.26
Czech Republic	0.36
Denmark	0.52
Estonia	0.37
Finland	0.37
France	0.27
Germany	0.31
Greece	0.29
Hungary	0.34
Ireland	0.36
Italy	0.42
Latvia	0.37
Lithuania	0.37
Luxembourg	0.40
Malta	0.27
Netherlands	0.40
Norway	0.54
Poland	0.34
Portugal	0.32
Romania	0.34
Slovakia	0.35
Slovenia	0.34
Spain	0.30
Sweden	0.52
Switzerland	0.36
United Kingdom	0.36

([9]) http://espreme.ier.uni-stuttgart.de.

The deposition velocity for mercury is much smaller than for other chemical species, at around 0.023 cm/s, owing to its long atmospheric residence time (one to two years). Mercury is a global pollutant. Global and regional estimates of the impact and damage cost of mercury air emissions (due to ingestion of methyl-mercury in contaminated fish products) have been carried out by Spadaro and Rabl (2008a).

A3.3 Pollutant transport and environmental fate analysis in soil and water

Environmental concentrations are calculated using the methodology developed by the US EPA for assessing multimedia transport in soil and freshwater bodies (US EPA, 2005). For the seawater compartment, the pollutant mass is computed assuming a first order process. Namely, the rate of change of mass in the compartment is equal to net change in the mass inflow and outflow. The outflow mass identifies the sink (pollutant settling to the bottom of the ocean), while the inflow mass is the source (mass flow into the ocean from freshwater bodies).

Environmental fate analysis comprises various stages:

- first, pollutant emissions to air;
- second, atmospheric dispersion and removal by deposition onto land and water surfaces or by chemical transformation;
- third, environmental accumulation, transport and estimation of concentrations in soil and water compartments;
- fourth, uptake by plants and animals;
- finally, passage through the human body on the way to its ultimate environmental disposal, which may involve, for example, soil fixation (the pollutant is trapped well below the surface layer in soils, making it no longer bio-available) or settling on water bed sediment.

At present, RiskPoll does not deal with discharges to water and soil, although the same methodology developed for air emissions may be extended to analyse these cases as well.

There are several routes of potential exposure to a pollutant, including inhalation, consumption of contaminated tap water, agricultural crops and animal products, such as fish, meat, milk, fruits and vegetables, and grains and cereals. All these pathways are addressed in RiskPoll. The inhalation dose depends very much on local conditions, especially the deposition velocity and the size of the population at risk. It contributes at most a few

Table A3.2 Ingestion dose by compound property

Compound property	As	Cd	Cr	Ni	Pb
Soil-water partition factor (m^3/kg_{Soil})	0.029	0.075	0.019	0.065	0.9
Suspended sediment-water partition factor (m^3/kg_{Sed})	25	2	50	100	200
Plant-soil bio-concentration (root uptake, kg_{Soil}/kg_{DW})					
Fruits and green vegetables	6.33E-03	1.25E-01	4.88E-03	9.31E-03	1.40E-02
Root vegetables	8.00E-03	6.40E-02	4.50E-03	8.00E-03	9.00E-03
Grains and cereals	4.00E-03	6.20E-02	4.50E-03	6.00E-03	9.00E-03
Animal feed	3.60E-02	3.64E-01	7.50E-03	3.20E-02	4.50E-02
Animal feed to cattle meat biotransfer factor (day/kg)	2.00E-03	1.20E-04	5.50E-03	6.00E-03	3.00E-04
Animal feed to cattle milk biotransfer factor (day/L)	6.00E-05	6.50E-06	1.50E-03	1.00E-03	2.50E-04
Freshwater fish bioaccumulation factor (L/kg_{Fish})	300	200	200	100	300
Seawater fish bioaccumulation factor (L/kg_{Fish})	1 000	1 000	200	1 000	200
Shellfish bioaccumulation factor (L/kg_{Fish})	2 000	20 000	800	2000	1 000

Sources (compound properties, human risk factors, and other useful information):
Human Health Risk Assessment Protocol (http://www.epa.gov/osw/hazard/tsd/td/combust/riskvol.htm#volume2);
Risk Assessment Information System (http://rais.ornl.gov/cgi-bin/tox/TOX_select?select=chem);
Integrated Risk Information System (http://cfpub.epa.gov/ncea/iris/index.cfm);
Adaptive Risk Assessment Modeling System (http://el.erdc.usace.army.mil/arams);
International Toxicity Estimates for Risk (http://www.tera.org/ITER);
Baes et al., 1984; IAEA, 1982, 1994 and 2001.

per cent of the total intake dose, but this does not imply that associated health impacts are negligible. The ingestion dose, on the other hand, is much less sensitive to local conditions because of food trade between different countries and regions. The ingestion dose is much more uniform than the inhalation dose (see Table A3.2).

Other avenues of exposure that are not addressed in RiskPoll include groundwater contamination, dermal contact and soil ingestion. Of these pathways, groundwater contamination could be of concern, but the remaining two items are usually negligible. Finally, it should be noted that the ingestion dose computed in RiskPoll represents a conservative estimate because no adjustment has been made to account for losses or reductions from food preparation and implementation of remedial strategies, other than specifying an efficiency of filtration for tap water consumption.

Table A3.3 Human and cattle dietary intake rates and population densities

Food consumption rates for European population (annual intake)			
	General population	**Infants (~ 1 % of population)**	
Drinking water (tap)	600	120	L
Fruits and above ground vegetables	88	86.3	kg_{FW}
Root vegetables	76	17.3	kg_{FW}
Grains and cereals	60	34.0	kg_{FW}
Beef meat	56	12.5	kg_{FW}
Fresh milk and other dairy products	101	275	L
Freshwater fish	3.6	0.32	kg_{FW}
Saltwater fish	6.0	0.55	kg_{FW}
Shellfish	1.8	0.21	kg_{FW}

Food consumption rates for beef and dairy cattle (daily intake)			
	Beef cattle	**Dairy cattle**	
Water intake	40	75	L
Forage	8.8	13.2	kg_{DW}
Silage	2.5	4.1	kg_{DW}
Grains	0.47	3	kg_{DW}
Soil ingestion	0.5	0.4	kg_{Soil}

Note: L = liters, kg_{FW} = kg of fresh weight, kg_{DW} = kg of dry weight, kg_{Soil} = kg of soil.

Sources: DAFNEsoft package (http://www.nut.uoa.gr/dafnesoftweb/), US EPA (2002 and 2005), IAEA (1994).

Population density estimates for an unknown source location in Europe

The continental population density is 80 persons/km², population averaged uniformly over land and water surface areas. This value is used for estimating the population total (collective) ingested dose, namely the total pollutant intake through diet. For the collective inhalation dose calculations, the regional population density is 112 persons/km². The exposed population is normalised by a surface area with a radius of 1 000 km, centred at the hypothetical source location. This value is a weighted average of country-specific population density estimates (see below).

Country-specific population density estimates for an unknown source location in that country					
Regional population density (persons/km²) varies by country of emission					
Austria	110	Germany	152	Norway	43
Balkans	73	Greece	55	Poland	97
Belgium	214	Hungary	106	Portugal	62
Bulgaria	53	Ireland	59	Romania	73
Cyprus	56	Italy	150	Slovakia	106
Czech Republic	116	Latvia	40	Slovenia	110
Denmark	83	Lithuania	52	Spain	55
Estonia	33	Luxembourg	138	Sweden	75
Finland	36	Malta	33	Switzerland	139
France	105	Netherlands	228	United Kingdom	122

A3.4 Impacts on human health

Pollutants that are carcinogenic via inhalation only include cadmium, chromium (valence state VI, which comprises roughly 20 % of chromium air emissions), nickel, 1,3 butadiene, diesel particulate matter, and formaldehyde. Inorganic arsenic, benzene, polycyclic aromatic hydrocarbon (PAH) compounds, such as benzo-a-pyrene (BaP), and dioxins/furans also act via the ingestion pathway. These pollutants are known human carcinogens. Oral exposure is particularly important for PAHs and dioxins/furans, contributing more than 98 % of the total impact on human health. Generally, oral exposure to inorganic arsenic accounts for about two thirds of the total damage cost. About 80 % of total arsenic in air is assumed to be inorganic, 50 % in tap water, 50 % in fruits and vegetables, and 25 % in grains (Schoof et al., 1999).

Lead and mercury (acting via methyl-mercury, MeHg, chemical transformation) are neurotoxins, which contribute to IQ loss in children, among other health impacts.

Inhalation unit risk factors [URF, lifetime excess cancer risk per μg/m³]

- 4.3E-3 for (inorganic) As, 1.8E-3 for Cd, 1.2E-2 for Cr-VI, 2.4E-4 for Ni, 3E-5 for 1,3 butadiene, 4.14E-6 for benzene, 3.37E-5 for diesel PM, and 1.3E-5 for formaldehyde.

Oral slope factors [SF, lifetime excess cancer risk per μg/(kgbw-day)]

- 1.5E-3 for (inorganic) As, 7.3E-3 for BaP, and 200 for dioxins/furans.

Dose response functions [DRF, Infant IQ loss per μg/day]

- 00.0416 for Pb and 0.036 for MeHg.

  Dose response relationships vary linearly with dose and do not have a 'no-effect' threshold value (i.e. impact is always positive for any intake dose).

Dose response functions [DRF, annual impact per person per μg/m³]

- DRF = URF/70 or DRF = SF/70 x ICf, assuming a lifetime exposure of 70 years. The parameter ICf is the intake to concentration factor; its value depends on the share of adult males and females and children in the exposed population (i.e. receptors), and on the mean breathing rates and body weights (kgbw) appropriate for each group of individuals. For Europe, ICf = 0.21 m³ per (kgbw-day). The population weighted mean breathing rate and mean body weight estimates are 12.6 m³/day and 64.3 kg, respectively. The mean breathing rate for an infant is 5.65 m³/day.

Sources: US EPA (1994, 1997 and 2002), Rabl and Spadaro (2006), Spadaro and Rabl (2008a), WHO (1999), IRIS (Integrated Risk Information System) database (US EPA, 2013b), and the NEEDS (2009) and MethodEX (2007) projects of the European Commission.

A3.5 Monetary valuation

Damage costs are calculated by multiplying the physical impacts (cancer cases or IQ points lost) by the appropriate unit cost (euros per incident). The default unit costs in RiskPoll are as follows (2005 EUR): EUR 2 000 000 for a fatal cancer, EUR 500 000 for a non-fatal cancer incident and EUR 9 300 for the loss of an IQ point. The cancer unit cost includes medical expenses (cost of illness), wage and productivity losses, and the willingness to pay to avoid the pain and suffering inflicted by the disease (welfare loss). Non-fatal cancers refer to incidents where the survival probability is greater than five years from the time of diagnosis. It is assumed that between 10 % and 20 % of cancer cases are non-fatal. The share is even greater for dioxins/furans, where up to 50 % of cancer cases are non-fatal. The unit cost of non-fatal cancers does not include welfare loss. The unit cost of an IQ point includes expenses associated with remedial learning and loss in potential lifetime earnings (Spadaro and Rabl, 2008a).

Costs are discounted at 3 % but without consideration given to increases in willingness to pay with economic growth in future years.

Table A3.4 Country-specific marginal damage costs for heavy metals, EUR/$kg_{emission}$ (based on RiskPoll, Ver. 2.0)

	Arsenic		**Cadmium**		**Chromium**		**Nickel**	
	Marginal damage cost	*68 % confidence interval*	**Marginal damage cost**	*68 % confidence interval*	**Marginal damage cost**	*68 % confidence interval*	**Marginal damage cost**	*68 % confidence interval*
Austria	345	*33–528*	27.5	*5–45*	36.7	*7–60*	3.7	*0.7–6.0*
Balkans	326	*31–499*	21.7	*4–36*	28.9	*5–47*	2.9	*0.5–4.7*
Belgium	407	*39–623*	47.0	*9–77*	62.6	*11–103*	6.3	*1.1–10.3*
Bulgaria	307	*29–470*	15.7	*3–26*	21.0	*4–34*	2.1	*0.4–3.4*
Cyprus	318	*30–487*	19.1	*3–31*	25.5	*5–42*	2.5	*0.5–4.2*
Czech Republic	347	*33–531*	28.2	*5–46*	37.6	*7–62*	3.8	*0.7–6.2*
Denmark	302	*29–462*	14.0	*3–23*	18.6	*3–31*	1.9	*0.3–3.1*
Estonia	282	*27–432*	7.8	*1–13*	10.4	*2–17*	1.0	*0.2–1.7*
Finland	284	*27–435*	8.5	*2–14*	11.3	*2–19*	1.1	*0.2–1.9*
France	365	*35–558*	31.0	*6–56*	45.4	*8–74*	4.5	*0.8–7.4*
Germany	393	*38–601*	42.4	*8–70*	56.6	*10–93*	5.7	*1.0–9.3*
Greece	309	*30–473*	16.2	*3–27*	21.7	*4–36*	2.2	*0.4–3.6*
Hungary	344	*33–526*	27.1	*5–44*	36.1	*7–59*	3.6	*0.7–5.9*
Ireland	303	*29–464*	14.3	*3–24*	19.1	*3–31*	1.9	*0.3–3.1*
Italy	355	*34–543*	30.7	*6–50*	40.9	*7–67*	4.1	*0.7–6.7*
Latvia	287	*27–439*	9.4	*2–15*	12.5	*2–20*	1.2	*0.2–2.0*
Lithuania	296	*28–453*	12.1	*2–20*	16.1	*3–26*	1.6	*0.3–2.6*
Luxembourg	353	*34–543*	30.2	*6–50*	40.2	*7–66*	4.0	*0.7–6.6*
Malta	292	*28–453*	10.8	*2–18*	14.4	*3–24*	1.4	*0.3–2.4*
Netherlands	417	*40–638*	50.0	*9–82*	66.7	*12–109*	6.7	*1.2–10.9*
Norway	279	*27–428*	6.9	*1–11*	9.2	*2–15*	0.9	*0.2–1.5*
Poland	335	*32–513*	24.5	*4–40*	32.6	*6–54*	3.3	*0.6–5.4*
Portugal	310	*30–475*	16.5	*3–27*	22.1	*4–36*	2.2	*0.4–3.6*
Romania	317	*30–485*	18.6	*3–31*	24.9	*5–41*	2.5	*0.5–4.1*
Slovakia	342	*33–523*	26.5	*5–43*	35.3	*6–58*	3.5	*0.6–5.8*
Slovenia	347	*33–531*	28.2	*5–46*	37.5	*7–62*	3.8	*0.7–6.2*
Spain	308	*29–471*	15.8	*3–26*	21.1	*4–35*	2.1	*0.4–3.5*
Sweden	297	*28–455*	12.6	*2–21*	16.8	*3–27*	1.7	*0.3–2.7*
Switzerland	364	*35–557*	33.4	*6–55*	44.6	*8–73*	4.5	*0.8–7.3*
United Kingdom	352	*34–539*	29.8	*5–49*	39.7	*7–65*	4.0	*0.7–6.5*

Note: Cost estimates (mean values) apply to air emissions, and include intake by inhalation and ingestion pathways. Generally, the ingestion dose tends to be uniform because of food transport between countries. Only carcinogenic impacts have been evaluated. The damage cost range assumes a lognormal distribution (Spadaro and Rabl, 2008b), with a geometric standard deviation of four for arsenic and three for the other heavy metals (presently, considered toxic only via the inhalation route).

Only inorganic arsenic and chromium in valence state VI are considered carcinogenic. About 80 % of total arsenic in air is assumed to be inorganic, 50 % in tap water, 50 % in fruits and vegetables, and 25 % in grains. Typically, 20 % of chromium air emissions occur as chromium VI.

Table A3.5 European marginal damage costs for heavy metal emissions to air (based on RiskPoll, Ver. 2.0)

	Intake fraction (ppm)	Health impact endpoint	Marginal damage cost (EUR/$kg_{emission}$)	*68 % confidence interval (EUR/$kg_{emission}$)*
Arsenic	890 (as arsenic) 160 (as inorganic As)	Cancer	**349**	*30–530*
Cadmium	2270	Cancer	**29**	*5.2–47*
Chromium	150	cancer	**38**	*7.0–63*
Lead	440 (entire population) 1.1 (infants only)	IQ loss	**965**	*90–1 480*
Mercury	870 (as mg Hg in methyl-Hg per kg Hg emission)	IQ loss	**910** (European estimate) **2 860** (global estimate)	*80–1 360* *240–4 290*
Nickel	550	Cancer	**3.8**	*0.7–6.3*

Note: The intake fraction is the amount of pollutant intake by the exposed population per unit emission rate. 'ppm' stands for 'parts per million', or equivalently, pollutant intake in mg per kg emission to air. Intake from inhalation is less than 1 % of the total.

Inorganic arsenic and chromium VI (about 20 % of chromium emissions) are carcinogenic. The share of inorganic arsenic varies by food product. About 80 % of total arsenic in air and 50 % in tap water is assumed to be inorganic. For lead computations, the infant intake fraction is the appropriate dose for estimating the IQ loss.

Mercury is a global pollutant, with a one- to two-year atmospheric residence time. The cost estimate for Europe corresponds to the impact (IQ loss) suffered by European citizens only. By contrast, the global assessment value applies to the worldwide population. The intake fraction is the mass of mercury (in mg) passing through the human body in the chemical form of methyl-mercury per unit air emission of mercury in kg.

Cost estimates (mean values) include intake by inhalation and ingestion pathways, and apply to a source of unknown location and characteristics (e.g. source stack height). Uncertainty intervals are based on a geometric standard deviation of 3 for cadmium, chromium and nickel, 4 for arsenic and lead, and 4.2 for mercury (Spadaro and Rabl, 2008a and 2008b).

Table A3.6a Country-specific marginal damage costs for organics, EUR/$kg_{emission}$ (based on RiskPoll, Ver. 2.0)

	1, 3 Butadiene		Benzene		PAH (as BaP equivalent)	
	Marginal damage cost	*68 % confidence interval*	**Marginal damage cost**	*68 % confidence interval*	**Marginal damage cost**	*68 % confidence interval*
Austria	0.49	*0.09–0.81*	0.075	*0.014–0.12*	1 279	*122–1 957*
Balkans	0.38	*0.07–0.62*	0.059	*0.011–0.10*	1 273	*122–1 948*
Belgium	0.82	*0.15–1.34*	0.120	*0.022–0.20*	1 296	*124–1 982*
Bulgaria	0.27	*0.05–0.45*	0.045	*0.008–0.07*	1 268	*121–1 940*
Cyprus	0.33	*0.06–0.54*	0.053	*0.010–0.09*	1 271	*122–1 945*
Czech Republic	0.49	*0.09–0.80*	0.074	*0.014–0.12*	1 279	*122–1 957*
Denmark	0.24	*0.04–0.40*	0.040	*0.007–0.07*	1 266	*121–1 938*
Estonia	0.14	*0.02–0.22*	0.026	*0.005–0.04*	1 261	*121–1 929*
Finland	0.15	*0.03–0.24*	0.027	*0.005–0.04*	1 261	*121–1 930*
France	0.59	*0.11–0.97*	0.088	*0.016–0.15*	1 284	*123–1 965*
Germany	0.74	*0.13–1.21*	0.109	*0.020–0.18*	1 292	*124–1 976*
Greece	0.28	*0.05–0.46*	0.046	*0.008–0.08*	1 268	*121–1 941*
Hungary	0.47	*0.09–0.77*	0.072	*0.013–0.12*	1 278	*122–1 955*
Ireland	0.25	*0.05–0.41*	0.041	*0.008–0.08*	1 267	*121–1 938*
Italy	0.53	*0.10–0.87*	0.081	*0.015–0.13*	1 281	*123–1 960*
Latvia	0.16	*0.03–0.27*	0.029	*0.005–0.05*	1 262	*121–1 931*
Lithuania	0.21	*0.04–0.34*	0.036	*0.007–0.06*	1 265	*121–1 935*
Luxembourg	0.52	*0.10–0.86*	0.079	*0.014–0.13*	1 281	*122–1 960*
Malta	0.19	*0.03–0.31*	0.033	*0.006–0.05*	1 263	*121–1 933*
Netherlands	0.87	*0.16–1.43*	0.127	*0.023–0.21*	1 298	*124–1 987*
Norway	0.12	*0.02–0.20*	0.024	*0.004–0.04*	1 260	*121–1 928*
Poland	0.42	*0.08–0.70*	0.066	*0.012–0.11*	1 276	*121–1 952*
Portugal	0.29	*0.05–0.47*	0.047	*0.008–0.08*	1 269	*121–1 941*
Romania	0.32	*0.06–0.53*	0.052	*0.009–0.08*	1 270	*121–1 944*
Slovakia	0.46	*0.08–0.75*	0.070	*0.013–0.12*	1 277	*122–1 955*
Slovenia	0.49	*0.09–0.80*	0.074	*0.014–0.12*	1 279	*122–1 957*
Spain	0.27	*0.05–0.45*	0.045	*0.008–0.07*	1 268	*121–1 940*
Sweden	0.22	*0.04–0.36*	0.037	*0.007–0.06*	1 265	*121–1 936*
Switzerland	0.63	*0.11–1.03*	0.094	*0.017–0.15*	1 286	*123–1 968*
United Kingdom	0.52	*0.09–0.85*	0.078	*0.014–0.13*	1 280	*122–1 959*

Note: Cost estimates (mean values) apply to air emissions, and include intake by inhalation and ingestion pathways. Generally, the ingestion dose tends to be uniform because of food transport between countries (for PAH, inhalation accounts for 2 % of total intake dose). Only carcinogenic impacts have been evaluated. The damage cost range assumes a lognormal distribution (Spadaro and Rabl, 2008b), with a geometric standard deviation of three for 1, 3 butadiene and benzene (presently, considered toxic only via the inhalation route), and four for the polycyclic aromatic hydrocarbons (PAH).

BaP = Benzo-a-pyrene.

Table A3.6b Country-specific marginal damage costs for organics (RiskPoll, Ver. 2.0)

	Diesel particulate matter EUR/$kg_{emission}$		Formaldehyde EUR/$kg_{emission}$		Dioxins/furans million EUR/$kg_{emission}$ (PCDD and PCDF)	
	Marginal damage cost	*68 % confidence interval*	**Marginal damage cost**	*68 % confidence interval*	**Marginal damage cost**	*68 % confidence interval*
Austria	0.56	*0.10–0.91*	0.21	*0.04–0.35*	27.0	*1.5–37.0*
Balkans	0.42	*0.08–0.69*	0.16	*0.03–0.27*	26.9	*1.5–36.8*
Belgium	0.92	*0.17–1.5*	0.35	*0.06–0.58*	27.3	*1.5–37.4*
Bulgaria	0.31	*0.06–0.50*	0.12	*0.02–0.19*	26.8	*1.5–36.7*
Cyprus	0.37	*0.07–0.61*	0.14	*0.03–0.24*	26.9	*1.5–36.8*
Czech Republic	0.55	*0.10–0.90*	0.21	*0.04–0.35*	27.0	*1.5–37.0*
Denmark	0.27	*0.05–0.45*	0.11	*0.02–0.17*	26.8	*1.5–36.7*
Estonia	0.15	*0.03–0.25*	0.06	*0.01–0.10*	26.7	*1.5–36.5*
Finland	0.17	*0.03–0.27*	0.06	*0.01–0.10*	26.7	*1.5–36.5*
France	0.66	*0.12–1.1*	0.26	*0.05–0.42*	27.1	*1.5–37.1*
Germany	0..83	*0.15–1.4*	0.32	*0.06–0.52*	27.2	*1.5–37.3*
Greece	0.32	*0.06–0.52*	0.12	*0.02–0.20*	26.8	*1.5–36.7*
Hungary	0.53	*0.10–0.87*	0.20	*0.04–0.33*	27.0	*1.5–37.0*
Ireland	0.28	*0.05–0.46*	0.11	*0.02–0.18*	26.8	*1.5–36.7*
Italy	0.60	*0.11–0.98*	0.23	*0.04–0.38*	27.0	*1.5–37.0*
Latvia	0.18	*0.03–0.30*	0.07	*0.01–0.12*	26.7	*1.5–36.6*
Lithuania	0.24	*0.04–0.39*	0.09	*0.02–0.15*	26.7	*1.5–36.6*
Luxembourg	0.59	*0.11–0.97*	0.23	*0.04–0.37*	27.0	*1.5–37.0*
Malta	0.21	*0.04–0.35*	0.08	*0.01–0.13*	26.7	*1.5–36.6*
Netherlands	0.98	*0.18–1.6*	0.38	*0.07–0.62*	27.4	*1.5–37.5*
Norway	0.14	*0.02–0.22*	0.05	*0.01–0.09*	26.7	*1.5–36.5*
Poland	0.48	*0.09–0.78*	0.18	*0.03–0.30*	26.9	*1.5–36.9*
Portugal	0.32	*0.06–0.53*	0.12	*0.02–0.20*	26.8	*1.5–36.7*
Romania	0.36	*0.07–0.60*	0.14	*0.03–0.23*	26.9	*1.5–36.8*
Slovakia	0.52	*0.09–0.85*	0.20	*0.04–0.33*	27.0	*1.5–36.9*
Slovenia	0.55	*0.10–0.90*	0.21	*0.04–0.35*	27.0	*1.5–37.0*
Spain	0.31	*0.06–0.51*	0.12	*0.02–0.20*	26.8	*1.5–36.7*
Sweden	0.25	*0.04–0.40*	0.09	*0.02–0.16*	26.8	*1.5–36.6*
Switzerland	0.71	*0.13–1.2*	0.27	*0.05–0.45*	27.1	*1.5–37.2*
United Kingdom	0.58	*0.11–0.95*	0.22	*0.04–0.37*	27.0	*1.5–37.0*

Note: Cost estimates (mean values) apply to air emissions, and include intake by inhalation and ingestion pathways. Only carcinogenic impacts have been evaluated. The damage cost range assumes a lognormal distribution, with a geometric standard deviation of three for diesel particulates and formaldehyde (presently, considered toxic only via inhalation), and five for the polychlorinated dibenzo-dioxins (PCDD) and dibenzo-furans (PCDF). For dioxins/furans, the inhalation exposure accounts for less than 2 % of the total intake dose. Generally, the ingestion dose tends to be uniform because of food transport between countries. Although the marginal damage cost for dioxins/furans is very high, the air emission rate is many orders of magnitude smaller than source emissions of the classical pollutants (e.g. primary PM and secondary aerosols) and the heavy metals (total cost = marginal cost * emission rate).

Table A3.7 European marginal damage costs for organic emissions to air (RiskPoll, Ver. 2.0)

	Intake fraction (ppm)	Health impact endpoint	Marginal damage cost ($EUR/kg_{emission}$)	*68 % confidence interval ($EUR/kg_{emission}$)*
1,3 butadiene	2.9	Cancer	0.50	*0.09–0.82*
Benzene	3.2	Cancer	0.076	*0.014–0.12*
PAH (BaP equivalent)	140	Cancer	1279	*120–1,960*
Diesel particulates	2.9	Cancer	0.56	*0.10–0.92*
Formaldehyde	2.9	Cancer	0.22	*0.04–0.36*
Dioxins and furans	160	Cancer	27 million EUR/kg	*1.5–37 million EUR/kg*

Note: The intake fraction is the amount of pollutant intake by the exposed population per unit emission rate. 'Ppm' stands for 'parts per million', or equivalently, pollutant intake in mg per kg emission to air. Intake from inhalation is less than 2 % of total (applies only to PAH and dioxins/furans).

Cost estimates (mean values) include intake by inhalation and ingestion pathways, and apply to a source of unknown location and characteristics (source stack height). Uncertainty ranges are based on a geometric standard deviation of three for 1,3 butadiene, benzene, diesel PM and formaldehyde, four for PAH, and five for dioxins/furans.

Annex 4 Sectoral adjustments

The methods used in this study recognise that the dispersion of emissions from point sources partly depends on characteristics specific to the emitting sector, such as stack height and flue gas temperature. Use of national average estimates of damage per tonne will introduce some error into the analysis if it ignores this issue. This Annex describes the methods used to adjust damage estimates for the main air pollutants by sector using the results of the Eurodelta II study (Thunis et al., 2008). This first requires conversion of the E-PRTR sectors to the Selected Nomenclature for sources of Air Pollution (SNAP) sectors used in Eurodelta II.

A4.1 E-PRTR to SNAP Conversion

Activities reported under the E-PRTR Regulation (EU, 2006) are grouped into nine categories:

1. energy
2. production and processing of metals
3. mineral industry
4. chemical industry
5. waste and waste water management
6. paper and wood production and processing
7. intensive livestock production and aquaculture
8. animal and vegetable products from the food and beverage sector
9. other activities.

Sector-specific correction factors developed under the Eurodelta II study (see Section 2.3) are applied to account for the differences in pollutant dispersion between specific sectors, as well as the all-sector averages computed through the available EMEP source-receptor matrices.

The emissions data analysed in the Eurodelta II study were reported in a different reporting format to the one used under the E-PRTR. In order to apply correction factors the facility/operator emissions need to be converted from the E-PRTR to SNAP format.

The E-PRTR categories, however, are more aggregated than SNAP. For example, the E-PRTR code 1C 'Thermal power stations and other combustion installations', referred to in this report as 'power generating facilities' covers:

- power stations (SNAP 1)
- commercial/public sector plants (SNAP 2)
- industrial facilities (SNAP 3).

Operators need to report their emissions under the E-PRTR at facility level. While facilities can report multiple activities, they must indicate their main activity. To illustrate, the reported emissions of NMVOC from Audi's facility at Ingolstadt were considered. This facility carries out three different activities (combustion, solvent use, waste disposal). The primary activity at the Audi factory was reported as 'combustion' (E-PRTR 1.1). However, it is probable that the NMVOC emissions are actually released from solvent use (i.e. painting of cars). Hence, based on the main activity, all of the NMVOC emission would be assigned to SNAP 3 (industrial combustion) rather than to SNAP 6 (solvent and other product use).

In total the E-PRTR database lists approximately 10 000 facilities for each reporting year. For the period 2008–2012, the E-PRTR contains information on releases to air from a total of 14 401 facilities. Due to the large number of sites under the E-PRTR, it is not possible to conduct a review of each facility and assign a SNAP code based on the different activities reported. The previous EEA report (EEA, 2011) described an analysis of E-PRTR versus SNAP classifications for the United Kingdom in 2008. It is acknowledged that the assumption that the majority of emissions by facility are associated with its main activity assignment does introduce an additional element of uncertainty. Nevertheless, it is believed that the overall sum for each SNAP code does still produce a representative estimate.

A4.2 Eurodelta II correction factors

The key results from the Eurodelta II report are presented in the following three tables. They show the ratio of 'sector efficiency' to 'all sectors efficiency' with respect to exposure of the European population to fine PM (health impacts of emissions of SO_2 and NO_X are estimated in terms of their contribution to sulphate and nitrate aerosols respectively) for emissions from France, Germany, Spain and the United Kingdom.

Where the ratio of sector efficiency to all sectors efficiency is less than one, control in the sector of interest is less effective in reducing population exposure per unit emission reduction than the average across all sectors. This tends to be the case for large industrial facilities, as typically tall stacks aid dispersion away from large centres of population. Where the ratio is greater than one, control in the sector of interest is more effective than the average, as is particularly the case for road transport ([10]).

In the case of SO_2, the relative efficiency of emission reductions for the public power sector is generally below 1 (Spain providing the exception) with an average of 0.87 and a range of ± 0.14. For the industrial sector values are in all cases close to 1 with a small range of ± 0.06.

In the case of NO_X, the relative efficiency of emission reductions for the public power sector is below 1 in all cases with an average of 0.78 and a range of ± 0.13. For the industrial sector, the average is 0.86 with a range of ± 0.07. For the road traffic sector, the value is greater than 1 in all cases, with an average of 1.12 and a range of ± 0.09. The absence of urban factors in the dispersion modelling will bias results significantly for this sector.

Table A4.1 Relative efficiency of sectoral SO_2 reductions for $PM_{2.5}$ impacts on Europe

Country	Sector efficiency/all sectors efficiency		
	1 Public power	3 Industrial	8 Other transport
France	0.74	1.06	
Germany	0.86	1.03	
Spain	1.01	1.03	1.06
United Kingdom	0.86	0.96	
Average	0.87	1.02	1.06
Range	± 0.14	± 0.06	

Table A4.2 Relative efficiency of sectoral NO_X reductions for $PM_{2.5}$ impacts on Europe

Country	Sector efficiency/all sectors efficiency		
	1 Public power	3 Industrial	7 Road traffic
France	0.91	0.87	1.05
Germany	0.80	0.84	1.06
Spain	0.65	0.93	1.15
United Kingdom	0.74	0.79	1.21
Average	0.78	0.86	1.12
Range	± 0.13	± 0.07	± 0.09

([10]) Results for the road transport sector are not of great relevance to this work as the sector is not included in the E-PRTR. However, they are included here to show how the reduction in transfer factors for sectors like public power relative to the all sector factors is balanced by increases elsewhere.

Table A4.3 Relative efficiency of sectoral primary PM reductions for $PM_{2.5}$ impacts on Europe

Country	Sector efficiency/all sectors efficiency				
	1 Public power	2 Industrial/ commercial	3 Industrial	4 Production processes	7 Road traffic
France	0.64	1.03	0.63	1.08	1.26
Germany	0.51	1.07	0.55	1.38	1.05
Spain	0.39	1.78	0.52	0.84	1.09
United Kingdom	0.47	1.04	0.58	1.31	1.51
Average	0.50	1.23	0.57	1.15	1.23
Range	± 0.14	– 0.20 to + 0.55	± 0.06	– 0.31 to + 0.23	– 0.18 to + 0.28

The level of variation for $PM_{2.5}$ impacts is greater than for SO_2 and NO_X, with average factors relative to the all sector efficiency being around – 50 % for sectors 1 and 3 and + 20 % for sectors 2, 4 and 7. For most sectors the variation around these averages is greater than 20 % at one or both ends.

Results from the preceding tables clearly show that there would be some level of error when applying an all-sector transfer factor. The most problematic of the three pollutants is primary $PM_{2.5}$ as its transfer factors depart from the all sector averages by a much greater degree than those for SO_2 and NO_X. However, available results from past external costs analysis suggest that emissions of $PM_{2.5}$ from most modern industrial facilities are sufficiently low compared to emissions of SO_2 and NO_X that this is likely to be of rather limited importance.

To investigate this, information from 141 analyses of the external costs of power plants in Europe of different designs and using different fuels were investigated. Fifty-seven of these cases could be considered relevant here ([11]). The external costs of NO_X and SO_2 combined outweigh those of $PM_{2.5}$ by an average factor of 14 (and a median factor of 6). This is despite the fact that the version of the ExternE methodology used gives higher weight to primary $PM_{2.5}$ than the CAFE benefits methodology. For only two facilities (both biomass) were the external costs of primary $PM_{2.5}$ estimated to be larger than those of SO_2 and NO_X combined.

A4.3 Limitations of Eurodelta II

In the course of the present study a number of limitations of the Eurodelta analysis have been identified, including those listed below.

1. Analysis focuses on emissions from only four countries. The representativeness of these countries is questionable. This could clearly generate uncertainty if the Eurodelta II results were extrapolated more widely across Europe. While it is understood that an additional four countries are to be considered in the near future, these data were not available for the present work.

2. One of the objectives of Eurodelta is to compare the results of different European-scale models. With this in mind, it was necessary to define a common modelling domain between the five models used in Eurodelta. The effect of this is to limit the overall area of the domain. A number of EU Member States and regions fall wholly or partially outside the modelled domain: Bulgaria, Cyprus, Estonia, Finland, Ireland, Malta (possibly), northern Scotland and much of Latvia, Lithuania, Romania and Sweden. Countries further east (e.g. Ukraine, Moldova and Russia) are also excluded. The results will therefore under-predict exposure to ozone and $PM_{2.5}$ ([12]).

([11]) The studies excluded from consideration here covered fuels for which emissions of $PM_{2.5}$ are very low or non-existent, such as nuclear, natural gas and most renewables (biomass excluded); small facilities that are not relevant to E-PRTR; and studies prior to 1998 (the time when chronic mortality impacts were brought fully into the ExternE Project methodology).

([12]) The results presented in the Eurodelta II report were derived using only one of the five models, understood to be the EMEP model. Results should therefore be available to extend the exposure assessment well beyond the Eurodelta II modelling domain. This would clearly require additional effort, either from the EMEP modelling team or from other teams that could process the EMEP-generated files. Were this to be done, the concern about limitation of the modelling domain would be very largely addressed.

3. No account is taken of enhanced urban exposures, though for the emission sources relevant to the E-PRTR this is unlikely to be of great importance.

4. The limitation of most importance may well relate to the treatment of stack height and the effective height of release. This appears to be discussed only in Section B5 of the Eurodelta II report (p. 96/106), which references a single Croatian report. It is not clear how representative the assumptions made here are of emission sources in the various SNAP sectors in the countries considered. Hence, while the assumptions made may be useful for demonstrating that there is an issue that should be addressed in analysis to support of policymaking, it is unclear how relevant the results of that modelling are to facilities across the EU, taking into account different attitudes to stack height calculation and (e.g.) different emissions linked to the use of different fuels. This is most important for SNAP sectors 1 (public stations), 3 (industrial plants) and 9 (waste) ([13]), which are those of most relevance to the E-PRTR.

A4.4 Approach adopted for this study

There are several ways of responding to the Eurodelta II results:

1. apply existing damage-per-tonne factors without adjustment for sector;

2. adjust by sector using the average of available sector/all sector transfer factors applied to all countries;

3. adjust by sector using country-specific sector and/or all sector transfer factors.

Option 1 would be followed if it were considered that the identified limitations were so great that they negated the value of the Eurodelta II results. However, while recognising these problems, it is logical that there will be some degree of sector-to-sector variation, and it would be better to take this into account than not to do so. At the other extreme, option 3 is only available for four countries, so could not be applied universally. An intermediate position has been taken, between options 2 and 3, applying country-specific data where they are available, and an average of country-specific factors for countries currently not covered by the analysis.

A4.5 Impacts of Eurodelta II on this study

The conclusions listed below were drawn from a review of Eurodelta II performed at the start of the present study.

1. Inter-sector variation for country-to-country pollutant transfer factors is significant, particularly for primary $PM_{2.5}$.

2. The method for estimating external costs should therefore be adapted to account for differences in transfer factors between sectors.

3. It is recommended that the work be started with a view to using the average sector-specific transfer factors from Eurodelta II, where country-specific factors are not yet available. There is sufficient consistency across countries for the sectors of most interest for the E-PRTR that associated errors should be manageable.

4. This position should be reviewed when further results become available that cover more countries.

5. The uncertainty associated with inter-sector differences is not great compared to some of the uncertainties that have been successfully addressed in past externalities work. It is also not great compared to the observed variation in transfer factors between the countries of Europe.

6. Ideally the sector-specific transfer factors would be calculated using the whole EMEP domain, rather than the restricted domain used in Eurodelta II.

([13]) SNAP sector 9 (Waste) was not considered in the Eurodelta II report.

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