Ex-post assessment of costs and benefits from implementing BAT under the Industrial Emissions Directive

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Abstract

This study has undertaken an ex-post assessment of impacts of implementing Best Available Techniques (BAT) conclusions (further referred to as BATC) for the iron and steel production sector to determine the level of environmental protection achieved and associated costs and benefits. A high-level assessment of the BATC impacts was undertaken but there was low confidence in the predicted impacts. A detailed bottom-up assessment was undertaken assessing each BAT individually using information gathered from industry and Member States and supplemented with data and assumptions. This exercise estimated the benefits of implementing the BATC outweighed the costs of implementing techniques by a factor of around 10.

The lessons learned from conducting the cost-benefit analysis for the iron and steel sector were used, together with a review of six further studies, to identify the most vulnerable parts of cost-benefit analysis methodologies.

Finally, linked to the appraisal of benefits of reducing pollution, the actions and activities required to develop damage costs at a national level have been described, together with information on the resources needed to do so, time required and timeframe, expertise and costs.

Cette étude a entrepris une évaluation à posteriori des impacts de l’implémentation des conclusions sur les meilleures techniques disponibles (MTD), ou CMTD, pour les secteurs de production du fer et de l’acier, afin de déterminer le niveau de protection environnemental obtenu ainsi que les coûts et bénéfices associés. Une évaluation de haut niveau sur les impacts CMTD a été faite mais la confiance dans ces impacts prédits fut considérée comme basse. Par une approche ascendante chaque MTD fut évalué en utilisant l’information recueilli chez le secteur industriel et le Etats membres et complétées par des données et hypothèses. Cet exercice a permis d’estimer que les bénéfices de l’implémentation des CMTD l’a emporté sur les coûts d’implémentation de ces techniques par un facteur 10.

Les enseignements tirés de la réalisation de l’analyse coûts-bénéfices pour le secteur du fer et de l’acier ont été utilisés, de même que l’examen de six autres études, afin d’identifier les éléments les plus fragiles des méthodologies d’analyses coûts-bénéfices.

Enfin, en liaison avec l’évaluation des avantages de la réduction de pollution, les actions et activités requises pour développer les coûts des dommages au niveau national furent détaillées, ainsi que les informations sur les ressources nécessaires pour leur mise en œuvre, le temps requis et les délais, le savoir-faire et les coûts.

Im Rahmen dieser Studie wurde eine ex-post-Analyse zur Umsetzung der besten verfügbaren Techniken (BvT) für den Sektor Eisen- und Stahlherstellung durchgeführt, um das erreichte Umweltschutzniveau und die damit verbundenen Kosten und Nutzen zu ermitteln. Zunächst wurden überschlägige Berechnungen zu den Auswirkungen der BvT-Schlussfolgerungen vorgenommen, wobei jedoch der Zusammenhang mit den erwarteten Auswirkungen gering war. Im Rahmen einer detaillierten Bottom-up-Analyse wurden die Kosten und Nutzen für jede einzelne BvT-Schlussfolgerung anhand von Informationen der Industrie und der Mitgliedstaaten sowie anhand von ergänzenden Daten und Annahmen ermittelt. Mit dieser Analyse wurde abgeschätzt, dass der Nutzen der Umsetzung der BvT Schlussfolgerungen gegenüber den Kosten für die Umsetzung der Techniken um einen Faktor von etwa 10 überwiegt.

Anhand der Erfahrungen aus der Kosten-Nutzen-Analyse für den Eisen- und Stahlsektor wurden zusammen mit einer Bewertung von sechs weiteren Studien die besonders anfälligen Einflussfaktoren von Kosten-Nutzen-Analyse-Methoden identifiziert.

Schließlich wurden in Hinblick auf die Bewertung der Vorteile der Verringerung der Umweltverschmutzung Maßnahmen und Tätigkeiten beschrieben, die zur Entwicklung von Schadenskosten auf nationaler Ebene erforderlich sind, zusammen mit Informationen über die dafür erforderlichen Ressourcen, den Zeitaufwand, das Fachwissen und die Kosten.

Executive summary

Introduction

This study has undertaken an ex-post assessment of impacts of implementing Best Available Techniques (BAT) conclusions (further referred to as BATC) for the iron and steel production sector in order to determine the level of environmental protection achieved and associated costs and benefits. In providing insights on the methodologies for assessing impacts of BATC, their feasibility and limitations, the report contributes to the lessons learned and recommendations made in the existing studies in that topic area.[1] It also provides guidance and an analytical framework for the Commission for carrying out and evaluating cost benefit analysis (CBA) when applied in the context of BATC.

Approach

First, a **scoping exercise** was carried out to identify BATs (for both air and water pollutants) considered to be the most important in terms of having the largest potential impacts on costs and benefits. This exercise identified the emissions sources, pollutants and BAT-AELs that were the focus of the subsequent assessments.

Second, a **high-level assessment** was carried out of what the emissions from the relevant iron and steel sector installations would be before and after implementing the I&S BATC. This high-level assessment only used publicly available data sources and sought to provide an estimate(s) of possible emissions impact using a simple method against which a more detailed assessment could be compared. The high-level assessment estimated using three methods.

1. Comparison of emissions reported in the period 2012-2016 with emissions projected to 2016 using reported emissions (in 2012) and reported changes in activity (production). The differences between the reported 2016 emissions and the projected emissions gives an indication of the impacts of BATC. This method is applied to both CLRTAP (Convention of Long Range Transport of Air Pollution) emissions data and E-PRTR emissions data.
2. Direct use of outputs from Amec Foster Wheeler (2015) where emission reductions were estimated for the sector using a combination of GAINS activity data and assumptions on technology uptake.
3. Comparison of IED BAT-AELs (2012) and limited values under the IPPC BREF (2001) to derive emission reduction factors for pollutants by process that were identified as high priority in the scoping stage. These emission reduction factors were then applied to estimated 2016 emissions (2012 E-PRTR projected using activity data) for the sector to estimate emission reductions.

Third, a **detailed process-level assessment of costs and benefits** was carried out. In-line with the findings of Ricardo (2016), this was a bespoke assessment for the sector in order to overcome data availability challenges. The assessment was based on available public and private datasets, and from data gathered from stakeholders. For the detailed CBA a database of steelworks was developed to underpin the analysis, aiming to capture a comprehensive list of all processes at iron and steel installations in the EU28. This used the PLANTFACTS dataset from the German Steel Institute (VDEh), and matched processes and installations with a list of ETS installations from the Emissions Trading Scheme Transaction Log as well as with facilities listed in E-PRTR.

The database was complemented by data from stakeholders. Integrated steelworks and standalone coke oven and pelletisation installations operators were contacted directly to participate in the study. An Excel-based questionnaire was developed to gather process-level information from the operators of

[1] Ricardo (2016) Analysis and development of methodologies for estimating potential industrial emissions reductions and compliance costs of BAT conclusions adopted under the Industrial Emissions Directive. *and*
AMEC Foster Wheeler (2015) Assessment of the potential emissions reductions delivered by BATC adopted under the IED.

integrated steelworks and coke ovens on whether changes were made to the installation and its processes as a direct result of needing to comply with the BATC, and if so, what those changes (investments or otherwise) were, when they occurred and if other drivers influenced the investments. The questionnaire also requested data on emissions and emission concentrations for a recent year prior to the changes due to BATC and a separate year post BATC compliance. The questionnaire was not targeted at operators of Electric Arc Furnaces (EAFs) due to the large number of EAF installations in the EU. Instead, Member State authorities were requested to provide information on the typical impacts of the BATC on EAFs.

Emission reductions resulting from compliance with the BATC were estimated using reported emissions data provided from the operators where possible. When this was not possible, two methods were used. First, and in most cases, an assumption on the technique abatement efficiency was used together with the estimated emission concentration achieved after applying the technique was fitted to back-calculate the emissions concentration prior to implementation.[2] Second, for a more limited number of cases, the changes in emissions were also estimated based on the reduction from an assumed existing emission concentration (the prevailing IPPC BAT-AEL) and the IED BAT-AEL, with an average taken where both methods could be used.

For assessing the costs needed to comply with BATC, information reported by installation operators has been used where available. Where this was unavailable, the costs of the specific techniques assumed were needed for compliance have been estimated using literature values of capital and operating costs (e.g. from the BREF), in-line with the methodology[3] in Ricardo (2016). These costs were usually scaled with capacity or production output. For monetising the benefits, the same approach as indicated[4] in Ricardo (2016) has been applied, using damage costs expressed in monetary units per tonne of emissions to air abated to estimate total health and environmental costs avoided. Capital costs were annualised over an assumed twenty year technique lifetime with a discount rate of 4% so that annual costs and benefits could be compared as a ratio.

Vulnerabilities of CBA methodologies for the assessment of BATC impacts

The lessons learned from conducting the CBA for the iron and steel sector have supported the identification of the most vulnerable parts of CBA methodologies. These conclusions have also been complemented by the review of nine studies (a mix of CBA and methodological papers) conducted using a common template designed to extract details of the methodologies and underlying assumptions. The overall objective was to identify those aspects of the CBA methodologies which have the largest influence on the final benefit-cost ratios and understand how these can vary in the assessment of BATC impacts. The information pulled out from the studies together with the outputs of the I&S CBA have been systematised for each step of the CBA (i.e. calculation of baseline emissions, emission reductions, costs, benefits). The vulnerability of various elements of the methodologies was then scored based on the evidence gathered in the study and expertise of the project team.

Deriving damage cost functions at national level

One of the highly vulnerable aspects of the CBA methodology is the assessment of benefits. The approach taken to the valuation of environmental and human health can greatly influence the final CBA results. One approach, also used in the CBA for the I&S sector in this study, is to apply damage cost functions to value benefits of pollution abatement. Yet, there are several aspects of the process of deriving the damage cost functions which can have large bearing on the resulting values. The study

[2] I.e. this is an analogue of the method marked as "ER3 Abatement Techniques" in Ricardo (2016), but as this is an ex-post assessment rather than ex-ante, the method relies on the latest available data on emissions or emissions concentrations after complying with the BATC in order to remove the need to account for changes in activity level between before and after BATC compliance when estimating the emissions impact.

[3] Labelled as method "CO2: Costing of additional investment (using CAPEX and OPEX) " in Ricardo (2016).

[4] Labelled as method "ABE2: Damage costs (EU, UK, DK)" in Ricardo (2016)

has investigated these aspects and how they differ across a sample of EU Member States. Based on this work, conclusions on the methodologies and resources required to derive damage cost functions at national level have been provided.

Findings

High-level estimate of emission reductions

The three high-level methods encompassed a wide range in the predicted impacts of the BATC on emissions, leading to low confidence in the predicted impacts.

The reliance in Method 1 on the use of E-PRTR means that the results are limited by the E-PRTR dataset itself. Over the reporting period 2012 to 2016, the number of iron and steel facilities reporting emissions data to E-PRTR drops, and for some air pollutants the proportional drop in numbers of facilities is as large or larger than the change between 2012 and 2016 emissions. This masked any potential analysis using this dataset of impacts on the BATC. The number of iron and steel facilities reporting emissions to water wasn't sufficiently large to enable a robust analysis.

Between the methods, there wasn't a consistent outcome that one Method estimated more extreme or more subtle changes; Method 2, which is arguably the most detailed method, for example estimates the highest PCDD/F impacts of all three methods, and the smallest Hg impacts of all three methods.

For emissions to air of SO_2, NO_X, dust and PCDD/F, Method 2 and Method 3 (comparison of IPPC and IED BAT-AELs) estimate larger emission reductions than are observed when comparing reported emissions with projected emissions (Method 1). However, for emissions to water, Method 1 predicts larger emission reductions due to BATC implementation for chromium, cyanides, lead, nickel and total nitrogen (not zinc) compared to Method 2.

Method 1 shows that in the case of many pollutants, reported emissions are lower than those projected. This is particularly the case when comparing 2016 projections to the reported emissions to air in E-PRTR. The analysis suggested there is a 27% reduction in SO_2, 13% reduction in NOx and a 25% reduction in emissions of Hg to air comparing reported emissions with those projected from 2012.

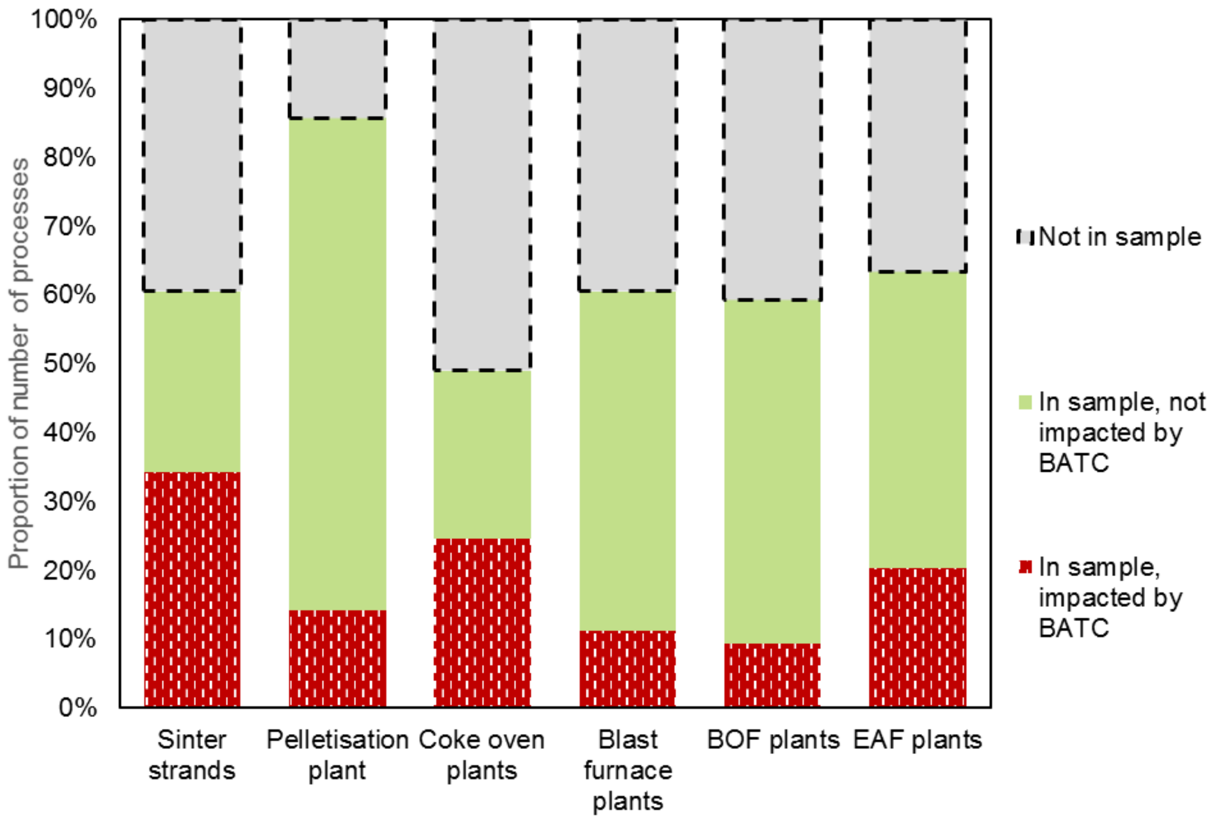
For Method 3, IED BAT-AELs are more stringent than IPCC BAT-AELs in most cases leading to anticipated emissions reductions. This includes 35% reductions in NOx emissions from coke ovens and 71% reductions from hot blast stoves, and 70% reductions in dust emissions from sinter plants. However, in some cases IPPC and IED BAT-AELs are identical, leading to no anticipated additional reductions in emissions with this method.

Detailed assessment of costs and benefits of complying with the BATC

The response rate of iron and steel operators to the questionnaire was overall low, with only 10 operators providing information. No information was available from operators on what techniques they would have fitted if the BAT-AELs applicable at their sites were the lower BAT-AEL range, so impacts of this could not be estimated. Most of the Member State authorities with steelworks responded to the request for information by providing high level data such as techniques installed to meet the BATC.

The processes that have been impacted the most by the I&S BATC, in terms of number of installations, are sinter strands, coke ovens and EAFs, according to the sample assessed (Figure E-1).

Figure E-1: Proportion of processes impacted or not by the BATC in the sample, and proportion not included in the sample due to lack of available information.



A summary of the estimated costs and benefits of fitting techniques to comply with the BATC is presented in Table E-1. Techniques installed to reduce dust emissions have been the most significant cost to operators. Compliance with BAT 20 (addressing primary dust from sinter strands) is estimated to have had the highest costs for operators among all BATs, with – for the sample of installations with information on impacts, which for sinter strands is 61% by number (13 out of 23) and 68% by capacity – total capital costs of €180 million, annualised to €13.2 m per annum, plus €18.6m per year operating costs. The most impacted process according to the CBA is sinter strands.

In addition to BATs for dust emissions, costs have been incurred to comply with BATs addressing other pollutants, including total annualised costs of €12.9m per year to reduce SO_2 from coke ovens and €4.9m per year to meet BAT for SO_2 from sinter strands.

Overall the estimated costs representing approximately two thirds of the industry on a capacity basis are €90m per year (total capital cost of €506m, and operating cost €52m per year).

Table E-1: Summary of costs and benefits.

Process	Number of processes impacted / in sample / in EU total	Total annualised costs (€m/yr)	Benefits (€m/yr) of BATC compliance			Benefit-cost ratio
			Processes with reported emissions	Remaining processes (estimated emissions)	Total for all processes impacted	
Sinter strands	13 / 23 / 38	45.9	242	402	644	14.0
Pelletisation plants	1 / 6 / 7	7.7	6.8	-	6.8	0.9 (Note 1)
Coke ovens	13 / 26 / 53	17.1	154	1	155	9.0
Blast furnaces	8 / 43 / 71	4.7	3.3	12.5	15.8	3.3
BOF	3 / 19 / 32	12.2	45.5	-	45.5	3.7
EAFs	40 / 125 / 197	2.0	2.1	63.4	65.5	33
TOTAL	**-**	**89.6**	**453**	**479**	**932**	**10.4**

Note 1: The results for pelletisation plants reflect a single Swedish installation. The benefit-cost ratio here is thus particularly sensitive to the PM damage cost for Sweden, which is about one quarter of the EU average.

The monetised health and environmental benefits arising from the emissions reductions are greater than the estimated costs of the techniques for compliance. Benefits from emission reductions from sinter strands are estimated to be €644 million[5], largely driven by the co-benefits on secondary pollutants when addressing dust emissions, leading to a benefit-cost ratio of 14 for sinter strands. Overall when factoring all processes in the sector the benefit-cost ratio is still very positive at 10.4.

Limitations

There were several limitations of the detailed CBA, the most important of which are as follows:

- It is unclear what costs would have been incurred under business as usual (i.e. a counterfactual scenario). The estimated costs are therefore likely overestimates of the true additional costs that would have been incurred beyond business as usual.
- Some of the investments made by operators may have been driven by factors other than the BATC, and thus attributing all of their costs to the BATC may overestimate the costs solely due to the BATC.
- Technique cost data has been taken from the BREF and in a few cases from installation operators. These data have been linearly scaled to other installations according to capacity or waste gas flow rate. Costs may not however scale linearly in practice.
- Annual benefits have been estimated from annual emission changes, which themselves have in many cases been estimated from daily average flue gas concentrations. To account for the uncertainty in the emissions estimates, two methods have been used (where possible) to estimate the emissions impacts.
- The damage costs used to estimate the benefits do not monetise all health and environmental effects of the emissions of certain pollutants, of short-time effects and of local situations, such that the estimated benefits may be underestimates of the true benefits.

[5] When including co-benefits, which for sinter strands make up ~72% of the total quantified benefits.

- The benefits have been estimated using a central value of a wide range of uncertainty in benefit valuation. The range represents the different approaches to valuation of VSL and VOLY. The benefits may thus be at least 50% higher or lower; cost/benefit ratios above 2.0 are more certain that benefits exceed costs. The damage costs used are specific to the Member State and taken from EEA (2014).

Conclusions from CBA of the iron and steel sector:

- **Data sources**: Ex-post assessment relies primarily on information being provided by operators and competent authorities, complemented by available collated public and private datasets. Few permits and permit decision documents were identified or made available, but those that were identified included useful information to support the CBA on installation capacity and techniques. National-level competent authorities provided useful higher-level data on the impacts of BATC but they lacked access to finer details on costs and emissions data which is only available at regional or local level. Use of a third-party database (VDEh) was necessary to estimate costs and benefits at the granularity of the BAT conclusions as public EU datasets such as E-PRTR are at installation and not process level.
- **Data availability**. More information is available on the impacts related to BATC on emission releases to air than emissions to water from operators and Member State authorities. Installations granted a derogation according to Article 15(4) should have more information available on the impacts of the BATC due to analysis conducted as evidence for the derogation requirement.
- **Stakeholder engagement**. Operators were generally reluctant to participate and provide data on impacts. For those installations where information was provided, it is difficult to validate the data as it is unique to the installation in question. It was difficult for operators to attribute investments costs solely to BAT compliance due to other drivers affecting their investment decisions. It is also difficult for operators to ascertain retrospectively what changes would have been made in the absence of the BATC (a hypothetical counterfactual scenario).
- **Results**: The detailed CBA has identified that a relatively small share of processes appear to have been impacted by the I&S BATC, which suggests that a high level of environmental protection was already being achieved and/or the BAT-AELs could have been set at a more stringent level. For those that were impacted, the benefits appear to outweigh the costs to a large degree. Secondary benefits of pollutants reduced in addition to the target pollutant have a large impact on the overall CBA.
- **Application to the Sevilla process**: the difficulties encountered for the CBA of the I&S BATC illustrates the impracticality of carrying out such an assessment as part of the Seville process. These challenges arise for multiple reasons including the individuality of installations, the inability of operators to distinguish BAT related costs and their unwillingness to provide information (which may be due to multiple reasons). For sectors other than iron and steel production, a similar CBA may be even more challenging, in cases where there are a larger number of installations in scope, and/or there is no additional supporting dataset available to assess impacts at the level of the BATC.

Vulnerabilities of CBA methodologies for the assessment of BATC impacts

The most vulnerable aspects of CBA methodologies identified through the CBA for the I&S industry and the studies reviewed are presented in Table E-2.

Table E-2: Key sources of vulnerability in CBA methodologies

Methodology aspect	Key sources of vulnerability
Overall approach and scope	**Time horizon** (i.e. the length of the appraisal period) can vary and is often influenced by the data available. Difference in the time horizon can lead to the streams of costs or benefits being omitted from the analysis, thus biasing the results. **Perspective** of the analysis, i.e. whether it is undertaken from a societal, private or other view, influences the coverage of impacts, selection of the data and overall approach. For example, societal CBAs will typically consider both costs and benefits, while CBAs by the private sector tend to focus primarily on costs to operators. **Operator coverage**, i.e. which sites are included in the assessment. Covering all operators is often not feasible and the choice of approach needs to balance available time, resources, and the size and variance of the sector.
Baseline scenario	Different **data sources** are available to different organisations and estimates for a single parameter in single year can differ widely in terms of its robustness, disaggregation and transparency.
Counterfactual scenario	None of the studies reviewed developed a counterfactual scenario suggesting that this is a less likely source of variation across the CBAs. This is perhaps because such forecasting introduces additional uncertainty into the assessment and is often difficult with available data.
Calculation of emission reductions	**The selection of air and water pollutants** will typically depend on the BATC of key concern, however general impact appraisal best practice dictates that analysts should identify all significant impacts. There can be large variation in pollutants coverage across the CBA studies. **Selection of techniques** (i.e. how abatement measures are chosen for application in the analysis) could also significantly impact on the results. Judging what technique is appropriate requires technical expertise and knowledge of the sector.
Calculation of costs	Whether a societal or private **discount rate** is applied. **Selection of unit technique costs** from underlying sources and whether modifications to these cost inputs are made compared to the original source. In practice, not many studies provide the necessary detail on how cost information is selected.
Calculation of benefits	**Coverage and the method taken** to assess the benefits of emission reductions, e.g. damage cost functions can account for health impacts only or capture also other benefits such as on buildings and ecosystems and crops. **The specifics of the assessment of health benefits drives variation** as impacts can be assessed at different levels of detail and disaggregation, using different tools.
Final results	A key vulnerability between studies is the acknowledgement of **uncertainty and sensitivity** and (if and) how this is explored. Uncertainty is inherent in all cost-benefit analysis, driven by the data inputs and methodologies available.

Moving towards best practice in damage costs estimation at national level

The quantification of national damage costs associated with pollutant emissions needs to be seen against several over-arching principles which have guided similar exercises in the past: the polluter pays principle, preventive and precautionary principles, multi-media and cross-boundary concerns, wider environmental and social goals and the EU's Better Regulation agenda.

The impact pathway approach (IPA) has been widely adopted for the quantification of health and environmental damage associated with the release of pollutants, and to derive damage costs. Estimation of damage costs following the IPA proceeds through several stages:

1. Quantification of emissions
2. Modelling of pollutant dispersion and chemistry
3. Exposure of people and sensitive environmental receptors
4. Quantification of impacts, using concentration-response functions
5. Valuation of quantified impacts
6. Review of uncertainties, including additional factors.

Estimating national damage costs requires access to particular technical expertise, including expertise in air pollution modelling, health impact assessment and economic valuation. It also requires access to specific tools and models, not least dispersion models which take account of a number of parameters in addition to the quantity of pollution released to depict the spread of emitted pollutants around the release site and the chemical transformations of pollutants leading to the formation of secondary species.

In each step of the IPA there are complexities and choices to be made in terms of approach, which will be guided by the time, resource and data available to the analyst. It is in these steps where the tools and data used can be flexed in order to define damage costs that are specific to a particular nation and/or sub-sector (e.g. industry).

Sets of damage costs already exist from which a range of best practices can be derived. This includes damage costs produced by the EEA which are provided at the national level already and hence are country specific. Existing damage costs vary in their pollutants covered, modelling of pollutant dispersion, coverage of impacts and the valuation of impacts.

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Abbreviations

BAT	Best available technique(s)
BAT-AEL	Best available technique associated emission level
BATC	BAT conclusion
BF	Blast furnace
BOD	Biochemical oxygen demand
BOF	Basic oxygen furnace
BREF	BAT Reference Document
CLRTAP	Convention on long range transboundary air pollution
COD	Chemical oxygen demand
COG	Coke oven gas
EAF	Electric arc furnace
EUTL	EU Transaction Log [EU ETS]
(EU) ETS	EU emissions trading scheme
GAINS	Greenhouse gas - Air pollution Interactions and Synergies model
I&S	Iron and steel
MS	Member State(s)
NACE	Nomenclature statistique des activités économiques (a sectoral classification system)
NFR14	Nomenclature for reporting (a sectoral classification)
PAH	Polycyclic aromatic hydrocarbons
PCDD/F	Dioxins: polychlorobenzodioxins (PCDDs) and polychlorodibenzofurans (PCDFs)
VDEh	Steel Institute: Association of German Steel Manufacturers
WWTP	Waste water treatment plant

1 Introduction

1.1 This report

This is the final report for project "*Ex-post assessment of costs and benefits from implementing BAT under the Industrial Emissions Directive*", which is Service Request 7 under framework contract ENV.C.4/FRA/2015/0042. The specific contract number is 07.0201/2017/761433/SFRA/ENV.C.4. The specific contract entered into force on 13 August 2017 and ran until 13 October 2018 (14 months).

1.2 Context

At an EU level, the primary instrument in place to mitigate the environmental impacts from EU industry is the Industrial Emissions Directive 2010/75/EU (IED). The IED entered into force on 6 January 2011. The IED combines and strengthens requirements previously set under 7 different Directives and was introduced to further control industrial pollution and lower the administrative burden associated with measures to reduce industrial pollution.

Following the approach of the IPPC Directive, the IED aims to ensure that emissions from different industrial sources are dealt with in an integrated way and minimised. All installations conducting activities listed in Annex I to the IED are required to operate according to a permit – issued by the relevant Member State authorities, and reflecting the principles and provisions stipulated by the IED. The permit extends to all environmental aspects of an installation's operating activities, including emissions, waste, resource use, noise, prevention of accidents and restoration of the site upon closure.

All permit conditions must be based on Best Available Techniques (BAT) conclusions (further referred to as BATC) within four years of adoption of the relevant BATC[6] which accompany the Reference Document (BREF). BATC can contain BAT-Associated Emission Levels (further referred to as BAT-AELs) (a numerical range of emission levels), BAT-Associated Environmental Performance Levels other than emission levels (further referred to as BAT-AEPLs) (e.g. BAT-AEPLs commonly concern raw materials, energy or water consumption, as well as waste generation) or may not be associated with either BAT-AELs or BAT-AEPLs (e.g. concerning monitoring, site remediation or environmental management systems).

Installations may apply for a derogation under Article 15(4) of the IED from BAT-AELs, where the installation operator can demonstrate that achieving the BAT-AELs would lead to disproportionately higher costs compared to the environmental benefits owing to the geographic location, local environmental conditions, or technical characteristics of the installation.

The BREF[7] for iron and steel production ("I&S BREF") was published in January 2013. The BATC[8] for iron and steel production ("I&S BATC") was published in March 2012 through "*Commission Implementing Decision of 28 February 2012 2012/135/EU establishing the best available techniques (BAT) conclusions under Directive 2010/75/EU of the European Parliament and of the Council on industrial emissions for iron and steel production*". The adoption of the I&S BATC concluded a six years long review process for the I&S BREF. The original BREF was adopted in 2001. Consequently, the competent authorities in the EU Member States had until March 2016 to update permits for the installations covered within the scope of the I&S BATC with the new requirements and for operators to

[6] With the exception of new installations that commence activities following the adoption of the BATC. These installations must comply immediately.

[7] http://eippcb.jrc.ec.europa.eu/reference/BREF/IS_Adopted_03_2012.pdf

[8] https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=OJ:L:2012:070:TOC

achieve compliance. The I&S BREF and BATC were the first to be published under the IED regime alongside those for the Manufacture of Glass sector.

There is a reporting requirement placed on Member States to report on the implementation of the IED over the period from 7 January 2013 to 31 December 2016 (Commission Implementing Decision of 12 December 2012, 2012/795/EU). The results from this reporting have not yet been assessed as submissions were delayed and only due to be submitted by June 2018. This reporting round included a 'sectoral spotlight' on implementation of the provisions in the BATC for the iron and steel sector and covered information on the status of updating permits for the installations, setting stricter permit conditions and different emission limit values than established in the BATC, details of the derogations granted according to Article 15(4) among others.

Fourteen sectoral BATC have now been published under the IED. During this period there have been discussions over the costs of compliance with BATC. Some Member States have advocated for the Commission to undertake *ex-ante* impact assessments of the proposals made as part of the BREF review process even though these are not required. Nevertheless, two studies have been commissioned by DG Environment focused on identification and implementation of potential methodologies to assess the impacts of the BATC proposals *ex-ante*:

- Amec Foster Wheeler (2015) "*Assessment of the potential emission reductions delivered by BATC adopted under the IED*"; and
- Ricardo (2016) "*Analysis and development of methodologies for estimating potential industrial emissions reductions and compliance costs of BATC adopted under the Industrial Emissions Direc*tive"

The key findings from Ricardo (2016) are shown in the box on the following page.

However, beside the major data limitations highlighted in the conclusions from both of these studies, *ex-ante* assessment faces a major difficulty in that it requires assumptions on how competent authorities would translate requirements of the BATC into permit conditions. Especially with regard to the BAT-AELs, stakeholders participating in the workshop organised as part of the Ricardo (2016) study raised the issue that it is impossible to predict whether for a given installation, emission limit values in the permits will be set at the higher or lower end of BAT-AEL range.

In addition, there are uncertainties over to what extent operators will request use of the Article 15(4) derogations, or in contrast whether competent authorities will set stricter BAT-AELs for some installations. A further study has been completed for DG Environment on the application of IED Article 15(4) derogations[9]. At the time of the assessment, 107 derogations had been applied for (of which 15 were from the Iron and Steel Production sector), and of these 107 applications 75 were granted. Of the 15 applications from iron and steel sector installations, 12 were granted, 1 was rejected and 2 were still pending at the time of the study.

Hence many IED stakeholders have expressed the view that a more accurate way to assess impacts of BATC is to evaluate them *ex-post*. With the four year deadline for updating the permit conditions for iron and steel installations across Europe now lapsed, there is an opportunity with the sector to evaluate the actual impacts of the BATC for this sector in terms of emission reductions, broader environmental benefits and costs.

[9] https://circabc.europa.eu/sd/a/9b59019b-df6c-4e6c-a5c2-1fb25cfe049c/IED%20Article%2015(4)%20Report.pdf

Box: Key findings of the Ricardo (2016) study recommendations on methodologies to assess emission reductions and compliance costs of BAT-AELs at the sector level

1. Each sector will need to have a bespoke approach to the assessment of impacts.
2. It is recommended that the first part of any assessment should be a scoping phase.
3. A framework for ex-ante assessment of the costs and benefits of BATC across all sectors should include the following steps as best practice:
    - Estimate baseline emissions based on reported emissions or concentrations where appropriate for the specific sector.
    - Estimate future emissions only if relevant changes are expected for the sector over time.
    - Estimate emission reductions that would occur if BAT-AELs were applied in parallel with the cost estimation, taking into account the impacts of specific techniques.
    - Estimate the costs of meeting BAT-AELs in conjunction with the emission reduction estimation by accounting for the costs of specific techniques needed to comply with BAT-AELs (using capital and operating costs).
    - Estimate benefits associated with the emission reductions if a case for action vis-à-vis as a comparison with costs is required. It is most straightforward to estimate benefits through the use of damage costs per tonne of pollutant abated for emissions to air. Non-monetised impacts should be considered. It was not possible to identify a method to quantitatively assess the impact of emission reductions on water quality.
    - Test sensitivity of the results against key assumptions
4. The best practices identified in recommendation 3 may need to be simplified to account for the available resources or time to undertake an assessment.
5. Technical and economic expertise is needed for all methodologies. Assumptions and calculations should be well-documented and transparent to allow review by stakeholders.
6. Accounting for the reality that there may be difficulties in obtaining large amounts of data required for a particular sector to carry out the recommended methodology, some alternatives/variants were identified. These include gathering smaller quantities of data for selected installations, and extrapolating the impacts of the BAT based on the assessment of impacts for these selected installations.
7. Data required for the assessments of potential industrial emissions reductions and compliance costs of BATC at sector level remain a key challenge. Advantages could be gained from:
    - Utilising Member States' reporting on implementation of the IED.
    - the development of a pan-European system to systematise all permit information across MS.
    - Using the data sought via questionnaires as part of the existing 'Sevilla' process.

1.3 Aims and objectives of the study

There were two principal aims of this study:

- **To undertake an ex-post assessment of impacts of implementation of BATC for the iron and steel production sector in order to determine the level of environmental benefit achieved and the associated costs**. This assessment will provide further insights on the methodologies for assessing impacts of BATC, their feasibility and limitations, thus contributing to the lessons learned and recommendations made in the existing studies in that topic area. The methodological findings will be presented to stakeholders, together with the quantitative cost-benefit findings.
- **To develop guidance and an analytical framework for the Commission for carrying out future analysis of the impacts of BATC and for the evaluation of cost benefit analyses.** This will be informed by the ex-post analysis described above and by the assessment of the vulnerabilities of methodologies for cost-benefit analysis when applied in the context of BATC, including how these methodologies differ between the Member States.

1.4 Scope and structure of this report

This report is structured as follows:

- Section 2 is the ex-post assessment of impacts of implementing the BATC for the iron and steel production sector.
- Section 3 is the assessment of the vulnerabilities of methodologies for cost-benefit analysis from other literature.
- Section 4 describes best practice in damage cost estimation at national level.

2 Assessment of the emission reductions (benefits) and costs of implementing the BAT Conclusions for the Iron and Steel Production sector

2.1 Aims and scope of the assessment

2.1.1 Aims

This task aimed to identify and quantify the impacts on installations due to implementing the I&S BATC. This includes the technology costs where changes were needed to comply with the BATC and the benefits associated with implementing these techniques due to the reductions in emissions released to the environment.

The assessment includes both those impacts that have already occurred (as the compliance deadline was March 2016), as well as those impacts that are in effect delayed due to derogations granted under Article 15(4) to installations.

2.1.2 Terminology

The terms 'installation' and 'process' are used as follows:

- Installation – used to refer to a single (integrated) steelworks comprising multiple processes. Three different datasets were used: a dataset of processes in steelworks (the VDEh database), the list EU ETS installations, and the facilities in E-PRTR. Consequently, the concept of installation is effectively close to the EU-ETS and IED concepts, but might differ from the permit situation.
- Process – A specific stage of the steel production process e.g. electric arc furnace or coke oven

For integrated steelworks the distinction between installation and process is clear with one installation comprising multiple processes, e.g. coke oven(s), blast furnace(s), sinter strand(s) etc. For some electric arc furnaces, pelletisation plants and standalone coke ovens that are not part of an integrated steelworks, one installation may only have one process i.e. an installation which is an electric arc furnace comprises one process of a single EAF.

2.1.3 Emission sources and pollutants in scope

The I&S BATC cover a range of emission sources and pollutants, and include BATCs that are generic in nature as well as process-specific BATCs both with and without BAT-AELs. This assessment is limited in scope to assessing the impacts on operators of those BATCs that have BAT-AELs. Furthermore, and in-line with the conclusions of Ricardo (2016), a prioritisation exercise was carried out to focus the assessment on those BATs that were considered to have the largest potential impacts on costs and benefits. This assessment was carried out mainly by focussing on BAT-AELs and by expert judgement of the project team member steel sector expert who was involved in the BREF process at Seville, by considering criteria as the expected impact of the BATC and the probability that reliable data can be provided, such as reliability of data from stationary sources vs. diffuse emissions.

A summary of the prioritised pollutants and emission sources that were agreed with the Commission to be in scope of this assessment are in Table 2-1. The full details of all the pollutants and emission sources that were considered for inclusion and exclusion for the assessment are shown in Appendix 1 with the motivation for their selection.

The prioritised pollutants in Table 2-1 differ only slightly from the scope of processes and pollutants in the previous literature: the iron and steel sector case study in Ricardo (2016) and from Amec Foster Wheeler (2015).

Table 2-1: Emission sources and pollutants in scope of the assessment per process

Process	Emissions source	Pollutant	BAT #
Sinter plant	Primary air emissions from sinter strands	Dust	20. BAT-AEL
		SO_2	22. BAT-AEL
		NO_X	23. BAT-AEL
		PCDD/F	24, 25. BAT-AEL
		Hg	21. BAT-AEL
	Secondary air emissions from sinter cooling and other relevant sources (discharge, crushing, screening, conveying)	Dust	26. BAT-AEL
Pelletisation plant	Air emissions from raw materials pre-treatment, induration strand, pellet handling and screening	Dust	33. BAT-AEL
		SO_2	34. BAT-AEL
		HCl	34. BAT-AEL
		HF	34. BAT-AEL
	Waste water	Suspended solids, Kjeldahl nitrogen, COD, heavy metals	39. BAT-AEL[10]
Coke oven plant	Coal grinding	Dust	42. BAT-AEL
	Coal charging	Dust	44. BAT-AEL
	Coke quenching	Dust	51. BAT-AEL
	Underfiring	NOx	49 BAT-AEL
		SO_2	49 BAT-AEL
	Desulphurisation of coke oven gas	Residual content of H_2S	48. BAT-AEPL
	Waste water	Sulphides, thiocyanate, cyanide, PAH, phenols, COD, BOD, ammonia-nitrogen	56. BAT-AELs
Blast furnace	Storage bunker of coal injection	Dust	59. BAT-AEL
	Cast house	Dust	61. BAT-AEL
	Hot stoves	Dust	65. BAT-AEL
		SO_2	65. BAT-AEL
		NO_X	65. BAT-AEL
	Waste water (Blast furnace gas treatment)	Suspended solids, cyanide, iron, lead, zinc	67. BAT-AEL
BOF plant	BOF gas recovery and cleaning - residual dust concentration	Dust	75, 76. BAT-AEPL
	Secondary de-dusting of BOF, including hot metal treatment, BOF-related processes and secondary metallurgy	Dust	78. BAT-AEL
	On-site slag processing	Dust	79. BAT-AEL
	Waste water (continuous casting and BOF gas cleaning if wet processes are applied)	Suspended solids, iron, zinc, nickel, total chromium, total hydrocarbons	80, 81. BAT-AEL
Electric arc furnace	Primary and secondary dedusting	Dust	88. BAT-AEL
		PCDD/F	89. BAT-AEL
		Hg	87. BAT, 88. BAT-AEL
	On-site slag processing	Dust	90. BAT-AEL
	Waste water (blast furnace gas treatment)	Suspended solids, iron, zinc, nickel, total chromium, total hydrocarbons	91-92. BAT-AEL

[10] Although waste water was initially excluded as only a small effect was expected, it was later included based on reporting by one MS.

2.2 Approach taken

The overall cost-benefit analysis was carried out in two phases:

1. A high-level assessment covering emissions impacts only, using publicly available data sources.
2. A detailed process level assessment of costs and benefits based on data gathered from stakeholders.

The approaches taken for each of these are in the following respective sections 2.2.1 and 2.2.2.

2.2.1 High level assessment of emissions impacts

The high-level assessment estimated what the emissions from the relevant iron and steel sector installations would be before and after implementing the I&S BATC. It is a theoretical exercise without a consultative element of data gathering. The purpose of this high level assessment was to:

a) provide estimate(s) of possible emissions impact using a simple method against which the more detailed assessment can be compared and assessed e.g. for accuracy.
b) contribute where possible to future scoping (prioritisation) of the processes within the iron and steel sector which are responsible for the highest emissions or where the highest emissions reductions due to implementation of BATC are expected

The focus of this high level estimate was on pollutants identified as high priority in the scoping stage of the assessment (section 2.1). Some pollutants were excluded from the analysis (suspended solids, COD, BOD, sulphides) as they are not reported in E-PRTR (TOC is however reported in E-PRTR).

Rather than selecting a single method for the high-level assessment, multiple methods have been considered to make high level estimates of the emissions impact of implementing BAT in the iron and steel sector. These methods are summarised as follows:

1. Comparison of reported emissions in 2012-2015 with emissions projected from 2012 to 2015 using reported changes in activity data, with the differences between the reported 2015 emissions and the projected emissions being an indication of the impacts of BATC. This method is applied to both CLRTAP emissions data and E-PRTR emissions data.
2. Direct use of outputs from Amec Foster Wheeler (2015) where emission reductions were estimated for the sector using a combination of GAINS activity data and assumptions on technology uptake.
3. Comparison of BAT-AELs under the IPPC BREF (2001) and those in the IED BREF (2012) to derive emission reduction factors for pollutants by process that were identified as high priority in the scoping stage. These emission reduction factors were then compared to emissions extracted from E-PRTR for the sector to produce emission reduction estimates.

The methods are described in detail in Appendix 2. The results from this assessment are in section 2.3.1. The estimates reflect the iron and steel activities considered in scope, which are IED activities 1.3, 2.1 and 2.2, and which are shown with E-PRTR and NFR equivalents in Table 2-2. National emissions to air data reported under the LRTAP Convention use the NFR codes, and are compiled at EU level from Member State submissions by the EEA. However, the NFR codes do not include capacity/activity thresholds for reporting as in the case of most IED activities, and so there is a risk of overestimation in using emissions reported under LRTAP when compared with IED/E-PRTR. However, the large steelworks are among the most polluting facilities in the EU, and so these are not expected to be sized below capacity/activity thresholds, such that the risk of overestimation of the LRTAP data is low, and discrepancies may instead be due to which category the emissions from associated activities – such as large combustion plants or ferrous metals processing plants – are reported under NACE division level corresponds to the sector level under the IED and to the emission thresholds of PRTR reporting.

In the iron and steel production sector, capacity thresholds are only relevant for IED activity 2.2. The capacity threshold of 2.5 tonnes per hour (i.e. around 20 kilotonnes per annum) seems to be more or less irrelevant, based on analysis of the capacities of EAFs in the VDEh database: only 8 EAFs are below this threshold, together making up less than 0.1% of the total EAF sector capacity. EAFs below the IED capacity threshold are expected to be very small specialised EAFs working under vacuum to produce high-purity steel. On the other hand, the emission reporting thresholds of E-PRTR certainly affect reporting of EAFs, and also limit the reporting of integrated steelworks for some pollutants.

Reporting by group under NACE bears some correlation to IED activities, including: manufacture of basic iron and steel and ferro-alloys (24.1); manufacture of tubes, pipes, hollow profiles and related fittings, of steel (24.2); manufacture of other products of first processing of steel (24.3); manufacture of basic precious and other non-ferrous metals (24.4); and casting of metals (24.5). NACE does not include capacity thresholds for reporting, meaning that reporting is more inclusive compared to the IED for certain activities (including IED activities 2.2 and 2.4) (Ricardo, 2018).

Acknowledging that integrated steelworks are very large sites encompassing multiple IED activities, their emissions reported under E-PRTR against an E-PRTR code for "main activity" may not therefore correlate with a single IED activity. For example, the Tata Steel integrated steelworks at IJmuiden in the Netherlands lists the following main and additional activities:

Main activity	Additional activities
1.(c) Thermal power stations and other combustion installations 2.(b) Installations for the production of pig iron or steel	1.(d) Coke ovens 2.(c).(i) Hot-rolling mills 2.(a) Metal ore (including sulphide ore) roasting or sintering installations 2.(f) Surface treatment of metals and plastics using electrolytic or chemical processes 5.(a) Disposal or recovery of hazardous waste

The consideration of the quality of mapping of datasets is also relevant in the case of activity data, which is used from the PRODCOM database in Eurostat. PRODCOM provides statistics on the production of manufactured goods, with 3,900 different products based on eight digit codes. The codes used for this analysis were for stainless, alloy and non-alloy steel produced in electric furnaces and processes other than electric arc furnaces. The main discrepancies in mapping is due once again to capacity/activity thresholds and due to the inclusion of associated activities in reporting under the IED and E-PRTR.

Table 2-2: IED activities in I&S BREF scope, with equivalent E-PRTR and NFR codes

IED activity in I&S BREF scope	Equivalent E-PRTR code	Equivalent NFR14 code
1.3. Production of coke	1.(d) Coke ovens	1 A 1 c Manufacture of solid fuels and other energy industries (*Note 1*) (1 B 1 b Fugitive emission from solid fuels: Solid fuel transformation)
2.1. Metal ore (including sulphide ore) roasting or sintering	2.(a) Metal ore roasting or sintering installations	2 C 1 Iron and steel production
2.2. Production of pig iron or steel (primary or secondary fusion) including continuous casting, with a capacity exceeding 2.5 tonnes per hour	2.(b) Installations for the production of pig iron or steel inc. continuous casting	2 C 1 Iron and steel production

Note 1: this NFR code 1A1c also may be used for IED activities 1.1 (combustion installations >50 MW) and 1.4 Solid fuel gasification and liquefaction) and therefore may not be a direct match to IED activity 1.3.

Note 2: NFR category 'Stationary combustion in manufacturing industries and construction: 1A2a Iron and steel' is not shown, as this is assumed to be reported as LCPs in IED activity 1.1.

The number of IED installations reported as permitted in 2015 in the EU28 for these activities, as assessed in Ricardo (2018) are:

- 10 installations covering activity 1.3 (coke ovens)
- 16 installations covering activity 2.1 (metal ore roasting or sintering)
- 22 installations covering activity 2.2 (production of pig iron or steel)

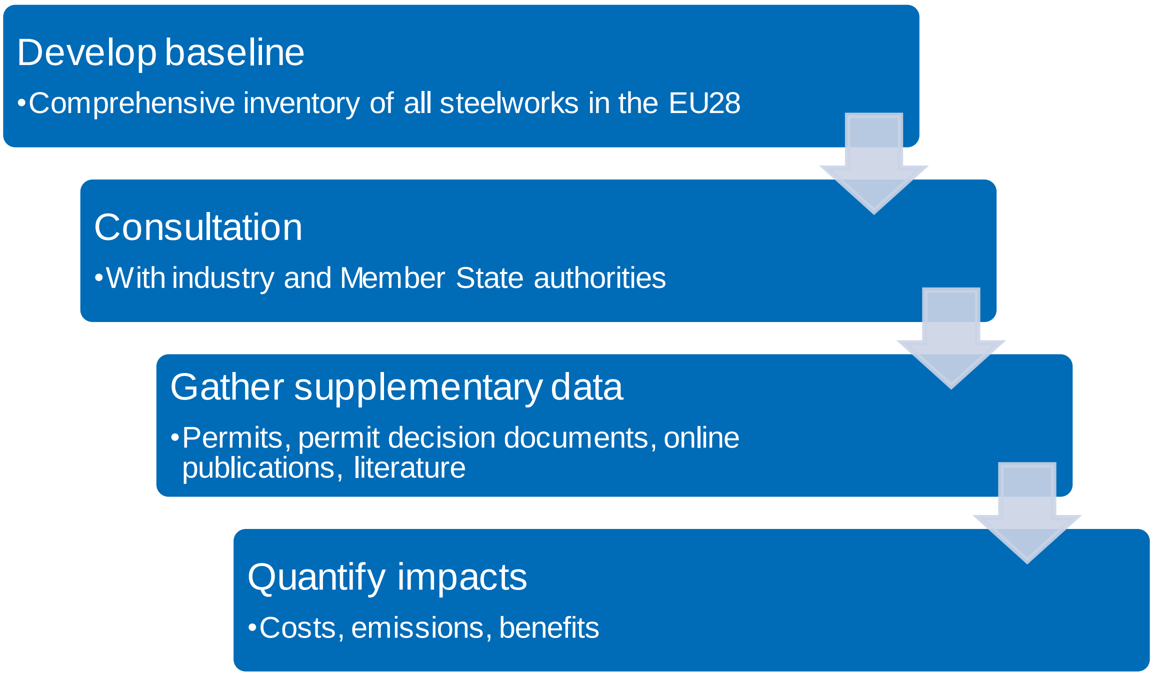
The above numbers are numbers of installations rather than numbers of processes as reported in section 2.3.2.1. It is not expected that all EAFs are included in the reported permitted activity data in Ricardo (2018) for activity 2.2.

2.2.2 Detailed installation and process-level assessment of costs and benefits from implementing the BATC

2.2.2.1 Summary of approach

The overall approach was to follow where possible the recommended methodologies described in Ricardo (2016) for carrying out cost-benefit analyses of BATC. However, as this assessment is an ex-post assessment (looking back rather than looking forward as described in Ricardo (2016)), some differences occur for the emissions calculations. The steps involved in carrying out the cost benefit assessment are summarised in Figure 2-2.

Figure 2-2: Steps in the cost benefit assessment



For assessing the **emissions impacts** resulting from compliance with the BATC, this work has aimed to rely on reported emissions data from operators where possible, for both before and after the changes at installations were made. In these cases where such data were available from operators, judgement was also needed as to whether it was necessary to account for activity level changes that occurred over the time frame between the provided years of data. For the cases where an impact has been reported, but no reported data on emissions were provided, the emissions impacts were estimated. The estimates were mostly based on the methodology recommended[11] in Ricardo (2016) which is, in conjunction with the cost assessment, using the estimated abatement efficiency of the specific technique identified as being deployed to comply with the BATC, and applying this abatement efficiency to an ex-ante projection of emissions *prior* to fitting the technique. Where this differs from an ex-ante assessment however is that, in the absence of reported emissions data, the emissions impacts have been estimated by applying the technique abatement efficiency to the estimated emission concentration achieved *after* applying the technique. For a minority of cases where the emission impact resulted as a secondary benefit from the application of a technique under another BAT, this approach was not taken, and instead the emissions have been estimated based on the reduction between an assumed existing emission concentration of the prevailing IPPC BAT-AEL and the IED BAT-AEL.

For assessing **costs** needed to comply with BATC, this work has aimed to rely on reported installation/process-specific cost data. Where this was unavailable, we have relied on estimating the costs of specific techniques using literature values of capital and operating costs, in-line with the recommended methodology[12] in Ricardo (2016).

For monetising the **benefits**, the same approach as indicated[13] in Ricardo (2016) has been applied, i.e. using damage costs expressed in monetary units per tonne of emissions to air abated to estimate total health and environmental costs avoided. This means no benefits of abating other emissions could be assessed.

2.2.2.2 Developing a comprehensive list of steel installations and processes in the EU28

A database of steelworks has been developed to underpin the detailed assessment, identifying processes and installations. The database aimed to include a comprehensive list of all the processes and installations in the EU28. A matching process has been carried out to link (map) the processes to installations i.e. resulting in knowledge for any process, which installation it belongs to, and vice versa for each installation what processes it includes. The sources of data used to compile the database in order of priority were:

1. The PLANTFACTS database purchased from the German Steel Institute (VDEh) in September 2017, of processes, which also includes the following characteristics for each process:
    a. Identification (operator, location, etc.)
    b. Operational status
    c. Capacity
    d. Abatement techniques (although not the performance / efficiency of these techniques)
    e. Year of modernising
2. Database of ETS installations from the EUTL of verified CO_2 emissions per year including installation names, operators and addresses
3. E-PRTR

The EUTL list was assumed to be comprehensive because of the prospect of fines for non-compliance with ETS reporting. In the cases of discrepancies, this EUTL dataset was therefore prioritised above the E-PRTR data for compiling a master list of installations.

The method for compiling the table of installations in the database was as follows:

[11] Labelled as method “ER3: Abatement techniques” in Ricardo (2016).

[12] Labelled as method “CO2: Costing of additional investment (using CAPEX and OPEX) " in Ricardo (2016).

[13] Labelled as method “ABE2: Damage costs (EU, UK, DK)” in Ricardo (2016)

- All installations that are implied by the VDEh database of processes were included, except where the VDEh database indicated that the processes or installations were mothballed or shutdown.
- Added to this were any installations EUTL list that were reported as iron and steel activity (labelled as activities 3, 4, 5, 22, 23, or 24 in the EUTL download) and which were not included in the VDEh database.
- Installations were excluded if any of the four following criteria were true:
    i. Installations with no verified CO_2 emissions for the three consecutive years 2014, 2015 and 2016
    ii. Installations with closed ETS registry accounts
    iii. Installations marked as iron and steel activity (activities 3, 4, 5, 22, 23, or 24) but which received no ETS allocation before 2013 and received allocation in the third period since 2013. These installations were assumed to be new in scope of the ETS from 2013 when it was extended to cover ferrous metal processing activities.
    iv. Installations which are, by inspection of their installation name, or by expert knowledge, understood to actually be ferrous metal processing installations, i.e. not under scope of the I&S BREF.

  As a final sense check, a manual inspection of the information available among the data sources and through online research was conducted. A final decision is then made as to whether the installation should or should not be included.
- The list established from VDEh and EUTL was checked against E-PRTR[14] using these rules:
    i. If a PRTR installation is not within the scope of the ETS, it was assumed not relevant, with the exception of installations listed under VDEh;
    ii. If a PRTR installation, which is also covered by the ETS has an ETS Activity other than 3, 4, 5, 22, 23, 24 or 25, it was considered not relevant.
    iii. If such an installation has an IED Activity other than 1.(d), 2.(a) or 2.(b) AND a NACE code other than 2410 AND it is not listed in the VDEh database, it was excluded.
- The draft list of installations and processes was checked with Member State stakeholders during the consultation elements described in the next subsections. This led to further removals of processes and/or installations that were known to have recently closed down, i.e. for which the impacts of the BATC are assumed not to have been felt (i.e. it was not considered that installations closed directly because of the BATC without other drivers).

2.2.2.3 Consultation with EU-level industry representation

EUROFER was contacted as part of this study, with the request to participate in the study. However, EUROFER formally declined to participate, citing competition compliance reasons and seeking to avoid handling sensitive data. Neither of these reasons would however have precluded EUROFER from encouraging its members to participate and contribute to the work.

Although EUROFER indicated (Personal Comm.) in this study that they did not carry out an ex-ante assessment of the cost impacts of the I&S BATC, earlier communication from EUROFER as part of the Ricardo (2016) study identified (based on Pers. Comm. with EUROFER 22 April 2016) the following:

[14] Carrying out the penultimate step relating to the E-PRTR database, three installations (2 German and one Polish installation) were identified as included in E-PRTR and in the VDEh database, but were not identified in the EUTL. The tap weight listed in VDEh suggested that these three installations are very small, which may explain why they are not covered by the ETS. Due to this low tap weight these three installations were excluded from the baseline database. Furthermore this step also identified 7 installations from the ETS list that could be excluded because the IED activity and NACE code are not related to the I&S sector and they were not covered by the VDEh database.

- The latest environmental data was collected from the individual operators for the BREF review between 2007-2012. Since then no information on the environmental performance or implementation status of the BATC has been collected at EU level by the industry.
- The competition compliance guidance limits the extent to which the detailed production / activity data for processes is shared or published in the public domain.
- At the time of the BREF review an estimation was done internally within EUROFER on the cost of the BATC to the sector. It was not published. Based on the information provided by the EUROFER membership to the EUROFER secretariat, it was roughly estimated the investment cost for the European Iron and Steel industry to be around 12 billion Euro to comply with the BAT-AELs listed in the I&S BATC.

EUROFER did not provide any further support to the study e.g. publicising the study nor did it encourage its members to participate.

2.2.2.4 Consultation with operators of integrated steelworks and standalone coke oven installations and pelletisation plants

Integrated steelworks as well as operators of standalone coke oven and pelletisation installations were contacted directly to participate in the study. The contact details for operators could not be provided by EUROFER and were sought through internet research and from telephoning general contact offices for the operators.

An excel-based questionnaire was developed to be used to gather information from operators. The questionnaire was limited to the scope identified in section 2.1. The questionnaire sought information at process level on:

- whether changes were made to the installation as a direct result of needing to comply with the BATC, and if so, what those changes (investments) were, when they occurred and what other drivers led to the investments.
- emissions and emission concentrations for a year prior to the investments due to BATC and a separate year post BATC compliance. The questionnaire was based on collecting information on the pollutants and media per process identified within scope as set out in section 2.1.

The questionnaire is included as Appendix 2. It was designed to be as streamlined as possible, with the use of conditional formatting to remove sections out of scope for an operator based on their answers to initial questions on their installation.

The operator questionnaires were circulated by email in January 2018 to 47 operator contacts representing 44 integrated steelworks and standalone plants and followed up with phone calls. The response rate of iron and steel operators to the questionnaire was overall low, with only 10 operators providing information. In summary the responses were:

- Ten installation operators indicated that they would not participate in the study.
- Four installation operators indicated that no investments had been made specifically due to the BAT Conclusions.
- Three installation operators provided questionnaires providing information on the impacts due to the BATC at their installations.
- Two installation operators stated that investments had been made but these were not due to compliance with the BATC (instead were due to other requirements of regional authorities and/or the availability of EU subsidies).
- One installation operator indicated that investments have been made but it is not possible to distinguish the extent to which the BATC was the primary driver for investment over other drivers.

Eleven installation contacts represented by 5 operating companies were not possible to contact and sixteen did not respond. No information was available from operators on what techniques they would have fitted if the ELVs applied at their sites were the lower end of the BAT-AEL range.

2.2.2.5 Consultation with Member State authorities

Due to the large number of electric arc furnace installations, operators of EAFs were not contacted directly. Instead, Member State authorities were approached to obtain information on the typical BATC compliance situation for Electric Arc Furnaces (EAFs). This approach was taken because there are many more EAF installations than integrated steelworks: 202 EAFs in the database compared with 29 integrated steelworks. The IED Forum contacts for each Member State were consulted to identify an iron and steel sector expert in the Member State that has knowledge and experience of permitting of steelworks across the Member State. These MS iron and steel sector experts were contacted directly to request to participate in the study. The Member States were also asked some questions relating to integrated steelworks.

A Microsoft Word-based proforma was developed to gather information from these Member State experts. The intention of the proforma was for it to be used either as a written questionnaire or as a guide during telephone interview discussions. As for the questionnaire on the integrated steelworks, the focus was on whether changes were typically needed at installations as a direct result of needing to comply with the BATC, and if so, the details of these changes. The proforma was based on collecting information on the pollutants and media identified as the scope in section 2.1.

The proforma is included as Appendix 3. It was tailored for each relevant Member State to include estimated numbers of EAFs identified in section 2.2.2.2.

The proformas were circulated by email in February 2018 after it was established that the conclusion of MS reporting of IED implementation, which includes a specific focus on the steel sector, should not be awaited due to postponement of the deadline for Member States. The response rates are shown in Table 2-3 below. No EAF installations exist in Cyprus, Denmark, Estonia, Ireland, Lithuania and Malta and these countries are therefore not included.

Table 2-3: Member State proforma responses

Member State	Responded to proforma	Type of response
Austria	Yes	In writing
Belgium	Yes	In writing and a follow-up interview
Bulgaria	No	-
Croatia	No	-
Czech Republic	Yes	In writing and a follow-up interview
Finland	Yes	In writing and a follow-up interview. Provision of permit documents
France	Yes	In writing
Germany	Yes	In writing and a follow-up interview
Greece	Yes	In writing
Hungary	Yes	In writing and a follow-up interview
Italy	No	-
Latvia	Yes	In writing and a follow-up interview
Luxembourg	Yes	In writing and a follow-up interview

Member State	Responded to proforma	Type of response
Netherlands	Indicated that no response could be given due to time constraints.	-
Poland	No	-
Portugal	No	-
Romania	No	-
Slovakia	No	Some information provided in an email
Slovenia	Yes	In writing and a follow-up interview
Spain	Yes	In writing and a follow-up interview
Sweden	Yes	In writing and a follow-up interview
United Kingdom	Yes	In writing and a follow-up interview

2.2.2.6 Drawing on literature for identifying impacts related to the BATC

In addition to information gathered from stakeholders, the following information sources were used:

- The VDEh PLANTFACTS database referred to in section 2.2.2.2
- Environmental permits of installations and permit decision documents
- The BREF
- Derogations
- Other information available online

VDEh PLANTFACTS database

The VDEh PLANTFACTS database includes, for some processes, additional information that is useful to give an indication on when an installation process was last upgraded and/or what techniques are used. This information was used in the determination of impacts in the following way: for installations that had a specific technique installed that was identified in the BREF as BAT, and the database indicated that the year that the process was last upgraded[15] was 2010 or earlier, it was assumed that no impacts were caused as a result of the BATC. On the contrary if the technique that is BAT was suggested to have been implemented since 2010, it was assumed that the technique was fitted as a result of the BATC compliance.

The techniques within VDEh which were taken as BAT are shown in Table 2-4. This table distinguishes between primary dust and secondary dust removal, which corresponds to primary emissions and secondary emissions of dust referred to in the I&S BATC. In particular it has been assumed that the installation of a bag filter complies with BAT in case of sinter plants and EAF (primary and secondary dust abatement) as well as for BOF plants (secondary dust abatement). Additionally for 5 sinter strands in the database, as ESP is permitted for compliance for existing plants in the BREF, this is assumed sufficient to not need bag filters. This assumption is however uncertain as not all ESP-equipped sinter strands may have met the BAT-AEL (or nationally implemented ELV) specified for ESP plants. In case of primary dust abatement at BOF plants it has been assumed that ESP, wet-type de-dusting and venturi scrubbers achieve BAT-AELs and BAT AEPLs, respectively.

[15] "Date of modernization" in the VDEh database.

Table 2-4: Abatement techniques reported in VDEh which were assumed to be BAT

Process	Primary dust abatement	Secondary dust removal
Sinter	Electrical Dry + Bag Filter Electrical Dry Bag filter	ESP + Bag Filter Bag Filter
BOF	Venturi Scrubber ESP Wet-type dedusting	Bag filter
EAF	Bag filter	Bag filter

Permits and permit decision documents

For a limited number of installations, their environmental permits implementing the BATC requirements were found to be publicly available online or were provided by MS experts and in some cases with supporting documents such as permit decision documents. The permit documentation identified were for installations in the Czech Republic, Finland, Romania, Slovakia, and the United Kingdom. For Sweden comprehensive environmental reports including pertinent parts of the permit and emission data were found to be publicly available or were provided by the Competent Authority.

These documents have generally provided information on the emission limit values applied, and techniques used at the installations. In those cases where an Article 15(4) derogation was being applied for or granted there was also information on technique costs and emission benefits. Of the permits assessed, details on costs and benefits due to the BAT Conclusions were found for two UK integrated steelworks.

BREFs

The 2012 I&S BREF was used to obtain technical information needed for the assessment, including:

- Technique costs and abatement efficiencies
- Flue gas flow rates
- Achieved emission levels associated with techniques

Derogations

Where information was available on installations that had been granted a derogation from BAT based permitting conditions, this was recorded in the model. Unless the derogation had no end date, installations / processes with derogations were assumed to install BAT by the expiry of their derogation period, and incur costs and benefits at this point. Installations with derogations were assessed as part of the overall BAT scenario but also separately.

Other information available online:

For some major upgrades in particular of sinter plants or coke ovens relevant information on costs and benefits could be found on presentations or other information publicly available online.

2.2.2.7 Calculating the costs of BAT impacts

Costs were calculated, where possible, for all processes and per BAT where an impact from the BATC has been identified. Costs were calculated based on responses from the operators and Member State authorities where available, and converted to price year 2015 using the World Bank GDP deflators for the EU28 (World Bank, 2018). These reported costs are shown in Table 2-5. Capital costs have been annualised using a 20 year economic life time, and a 4% discount rate (interest rate).

In cases where reported costs were not available, costs were estimated using two possible methods:

- Capital and/or operating costs were taken from the BREF. These were expressed for example in EUR/t of output or EUR/Nm^3/h. These were converted to 2015 price year. These data are included in Table 2-6.
- In cases where reported costs were available for another similar installation (e.g. in the same Member States), those reported costs were converted to costs per tonne capacity and then this cost rate was applied to another installation using its capacity data.

The costs of the remaining economic life of existing techniques installed at installations was not deducted from the calculated costs, nor were the costs of business-as-usual upgrades to existing installed techniques. Whilst these would ideally be taken into account, such approaches require robust data to be available widely on the existing techniques installed and the years when they were installed.

In some cases it wasn't possible to quantify both the costs and/or the benefits for an impact on an installation. A list of these cases is described by BAT number in section 2.3.

Operator-reported costs are taken to be accurate, although they may be over or under-estimated, it is not possible to verify them other than through comparison such as with costs derived from the BREF. As noted in Section 3.2, cost data reported by installations is difficult to verify as it is ultimately the result of site-specific factors and may also be based on confidential information. One way of attempting to verify reported costs is through comparison with publicly available data sources. In Table 2-5, where possible reported costs are displayed next to costs from the iron and steel BREF.

Table 2-5: Reported technique costs

BAT number	Technique	Price year	Capital cost quoted (€m)	Specific capital costs (€/t)	Operating cost quoted (€m/yr)	Specific operating costs (€/t/yr)	Country, Source, Installation
20	Bag filter/Activated carbon injection	2008	21.2	7.5	5.05	1.8	Operator 1
20	Bag filter	2011	22.7	10.3	No information	No information	Operator 2
22	Injection of adsorption agent	2011	1.4	0.5	4.8	1.72	Operator 1
26	Secondary dedusting: Bag filter	2016	7	2.5	0.03	0.01	Operator 1
42	Efficient extraction and subsequent dedusting (Coal grinding)	2015	0.135	0.097	0.01	0.072	Operator 1
42	Bag filter	2016-2018	1.53	2.6	No information	No information	Operator 3
44	Efficient extraction and subsequent dedusting (Charging car)	2014	2.4	1.75	0.04	0.029	Operator 1
48 & 49	Coke oven gas desulphurisation	2015	27.1	29.16	5.4	8.06	Member State permit
48 & 49	Coke oven gas desulphurisation	2016	34 (for 2 coke ovens)	50	No information	No information	Member State permit
51	Quenching tower	No information	22	17	No information	No information	Press release
51	Coke quenching abatement	2016	12.3 (for 2 coke ovens)	18.08	No information	No information	Member State permit
59	Secondary de-dusting – ESP/bag filter	2013	0.8	0.5	0.05	0.03	Operator 1

BAT number	Technique	Price year	Capital cost quoted (€m)	Specific capital costs (€/t)	Operating cost quoted (€m/yr)	Specific operating costs (€/t/yr)	Country, Source, Installation
61	Capture dust emissions and subsequent dry dedusting – cast house dedusting upgrade	2016	No information	No information	0.14	0.05	Operator 1
78	Bag filter	2016	16	10.2	1	0.63	Operator 4
80&81	Suspended solids: Recirculation of cooling water and water from vacuum generation	2015	8.7	5.5	No information	No information	Operator 4
88	Bag filter upgrade	No information	0.08	No information	No information	No information	Member State proforma, regional authority
88	Bag filter upgrade	No information	0.05	No information	No information	No information	Member State proforma, regional authority
89	High temperature quenching system	2015	1.2	1	No information	No information	Member State proforma, regional authority
89	Powdered activated carbon injection	Estimate – planned investment	0.6	1.3	0.3	0.6	Member State proforma, regional authority
89	Activated carbon injection	2016	0.15	0.18	0.1	0.125	Member State proforma, regional authority
89	Activated carbon injection	2015	0.01	0.01	0.08	0.1	Member State proforma, regional authority

Note: Cost data taken from the BREF has been uplifted to price year 2015 from 2012 the year in which the BREF was published. However, this cost data dates back to 1997 in most cases. The costs have not been inflated from 1997 on the reflection that inflation is balanced by the learning rate in technique manufacturers, where innovation and increase demand for equipment lead to price changes in the opposite direction to inflation

Table 2-6: Technique abatement efficiencies and costs – assumptions used where an impact is identified by the technique but the cost/emissions impact is unknown.

BAT	Technique	Pollutant abatement efficiency (%)	Price year	Capital costs (€/t)	Capital costs (€ /1000 Nm^3 / h)	Operating costs (€/t)	Operating costs (€/1000 Nm^3/h)	Basis / source for assumption
20	Bag filter	99% (dust)	2013	No information	16-35	0.3-0.6	No information	BREF p.127-139; 496
20	ESP (assumptions needed for existing plants)	95% (dust)	1996	Not applicable	Not applicable	Not applicable	No information	BREF p.119-125; 491
24&25	Lignite injection	No information	2010	No information	9.53	No information	No information	BREF, p.265
26	Secondary dedusting : Bag filter	No information	1996	12	No information	3.25	No information	BREF p161-162; 497
33	Bag filter (including reheating equipment)	99% (dust)	2007	5.43	No information	1.7	No information	BREF, p.196
42	Bag filter	99% (dust)	2016-2018	2.6	No information	No information	No information	Operator 3
48&49	Wet oxidative desulphurisation	No information	2016	No information	No information	50	8.9	BREF p.319-322; 509
48&49	Wet oxidative desulphurisation unit (Stretford process)	No information		30	300 000 per oven	No information	No information	BREF
51	Quenching tower	No information	Unknown	17.7	No information	No information	0.065	Press release
51	Coke dry quenching abatement	No information		100	5	No information	No information	BREF, p268-270; 506

BAT	Technique	Pollutant abatement efficiency (%)	Price year	Capital costs (€/t)	Capital costs (€ /1000 Nm^3 / h)	Operating costs (€/t)	Operating costs (€/1000 Nm^3/h)	Basis / source for assumption
61	Cast house dedusting	No information	1996	4.83	No information	0.32	No information	BREF, p.322, and Eurostat for energy costs
61	Cast house dedusting	>90%	2016	No information	No information	0.14	50.3	Operator 1
78	Bag filter	99% (dust)	2016	10.2	No information	0.63	No information	Operator 1
88	Bag filter	No information	Unknown	0.5[16]	No information	No information	No information	BREF, p. 47-0

[16] It is not clear whether the cost information relates to the entire abatement installation or only to the equipment related to PCDD/F abatement (BAT 89)

2.2.2.8 Calculation of change in emissions

Changes in emissions have been evaluated per process at BATC level.

Using actual reported emissions data

Emissions of installations from both before and after BATC techniques were implemented were provided in a limited number of cases from the operators in questionnaires.

In these cases where such data were available from operators, judgement was also applied as to whether it was also necessary to account for activity level changes that occurred over the time frame between the provided years of data. For example, for the Member State of one of the operators reporting emissions, the production volumes of steel as reported in World Steel from this Member State's integrated steelworks did not show any real change over the period between the emissions measurement, such that in this example the changes in production levels were assumed to not be important for consideration of the emissions under the counterfactual scenario.

Estimating the emissions change for a known installed technique.

In cases where a change in the abatement due to the BAT has been reported, but emissions were not reported, emissions changes resulting from implementing techniques were estimated where possible. The method of estimation used depended on the level of data available. Two methods were used; in some cases both methods are available for the same process, allowing a measure of the uncertainty to be estimated. Both methods rely on needing to convert emissions concentrations to mass emissions (see below).

Method 1: The estimates were mostly based on the methodology recommended[17] in Ricardo (2016) which is, in conjunction with the cost assessment, using the estimated abatement efficiency of the specific technique identified as being deployed to comply with the BATC. Abatement efficiencies were reported in some cases but in most instances were taken from the BREF.

In this method, the emissions concentration reported or calculated after abatement were used, and from that, an estimate of what emissions would have been (i.e. counterfactual) before installing the technique by applying the assumed abatement efficiency of the technique. For example, the emissions concentration to be achieved by a technique after being fitted would be taken from the BREF or taken to be the upper BAT-AEL.

In some cases, the technique being fitted as a result of BATC compliance was replacing an existing technique (e.g. bag filter replacing ESP for sinter strands). In these cases, the abatement efficiency of the existing technique also had to be assumed in order to estimate only the incremental change in emissions from switching from the less efficient to the more efficient technique. In this same example, replacing an ESP that was assumed to abate dust at 95% with a bag filter that was assumed to abate dust at 99% efficiency leads to an effective incremental abatement efficiency of: 1- [(1-99%)/(1-95%)] = 80%.

Method 2. An alternative method of estimation was also used in instances where both IPPC BAT-AELs and IED BAT-AELs exist for a given pollutant and process combination. In these cases the annual mass emissions were calculated from each of the given limit values to estimate theoretical emissions before and after the BATC, and the difference between them was the emissions savings. Expert judgement was used in some cases where BAT AELs were deemed to have not been an accurate representation of real world performance. The prominent example of this is for sinter strands BAT 20, where the upper IED BAT AEL was judged by the team to be higher than actual concentrations achieved by operation with bag filters. As such a pre-abatement emission level of 50 mg/Nm^3 and a post-abatement efficiency of 6.1 mg/Nm^3 was assumed, based on figures reported in the BREF.

[17] Labelled as method "ER3: Abatement techniques" in Ricardo (2016).

Converting between concentrations (mg/Nm^3) and mass emissions (kg/yr)

For both of the above-mentioned methods, concentrations were converted to annual mass emissions using reported/estimated flow rates and assumed annual operating hours of 8,500. As integrated steelworks operate year-round, an annual total hours of operation of 8,500 hours was chosen to reflect this with some downtime for maintenance. The use of daily average concentrations such as limit values from the BREF to estimate annual emissions effectively assumes that the daily mean limit value is met throughout the year; implicitly that the iron and steel industry is not subject to seasonal fluctuations.

Where flow rates were not reported by the operator it was possible to derive the volume flow from the VDEh database for sinter plants and for EAFs. Where flow rates were indicated in the VDEh database as nominal output of the suction fan at the reported temperature volume flow has been converted to standard conditions (Nm^3/h; temperature 273.15 K, pressure 1.02315 bar) by assuming an ideal gas.

In the case of EAFs it was observed that there were some outliers in the maximum fan capacity field in VDEh where the ratio of flow rate to capacity was very low; in cases where the ratio was lower than 0.5 the trend was to use the maximum fan capacity rather than the database value.

In instances where flow rate was not reported or not possible to be calculated, emissions were estimated by modifying reported emissions from another installation where appropriate using capacity. In instances where concentrations were not reported, IED upper BAT-AELs were used as the assumed concentration post-abatement.

In these estimation methods, changes in activity in the period are not accounted for. This is because installation level activity data was not available. Analysis of Member State level steel production from 2012 to 2016 is provided below for illustrative purposes for both steel produced from oxygen furnaces (Table 2-7 and Figure 2-3) and steel produced from EAFs (Table 2-8 and Figure 2-4). Although decreases in emissions from reduced production may therefore exist, for most Member States there has been steady or small increases in production of BOF steel between 2012 and 2016, and steady or small decreases in production of EAF steel between 2012 and 2016.

Table 2-7: Steel production from oxygen furnaces in the EU 2012-2016 (kt) (source: World Steel)

	2012	2013	2014	2015	2016
Austria	6746	7288	7185	7020	6766
Belgium	4647	4738	4952	4809	5330
Czech Republic	4701	4805	5006	4902	5011
Germany	28872	29185	29881	30054	29486
Finland	2300	2220	2545	2625	2750
France	9507	10195	10645	9825	9527
Hungary	1488	744	974	1507	1041
Italy	9312	6798	6514	4791	5669
Netherlands	6739	6581	6839	6888	6824
Poland	4234	4399	5067	5321	5110
Romania	1701	1830	1845	2245	2228
Slovakia	4023	4172	4344	4236	4506
Spain	3423	4210	4206	4701	4547
Sweden	2883	2986	3096	2890	3111
United Kingdom	7525	9915	10165	9051	6153
EU	98103	100067	103265	100864	98058

Figure 2-3: Steel production from oxygen furnaces in the EU 2012-2016 (2012=1) (source: World Steel)

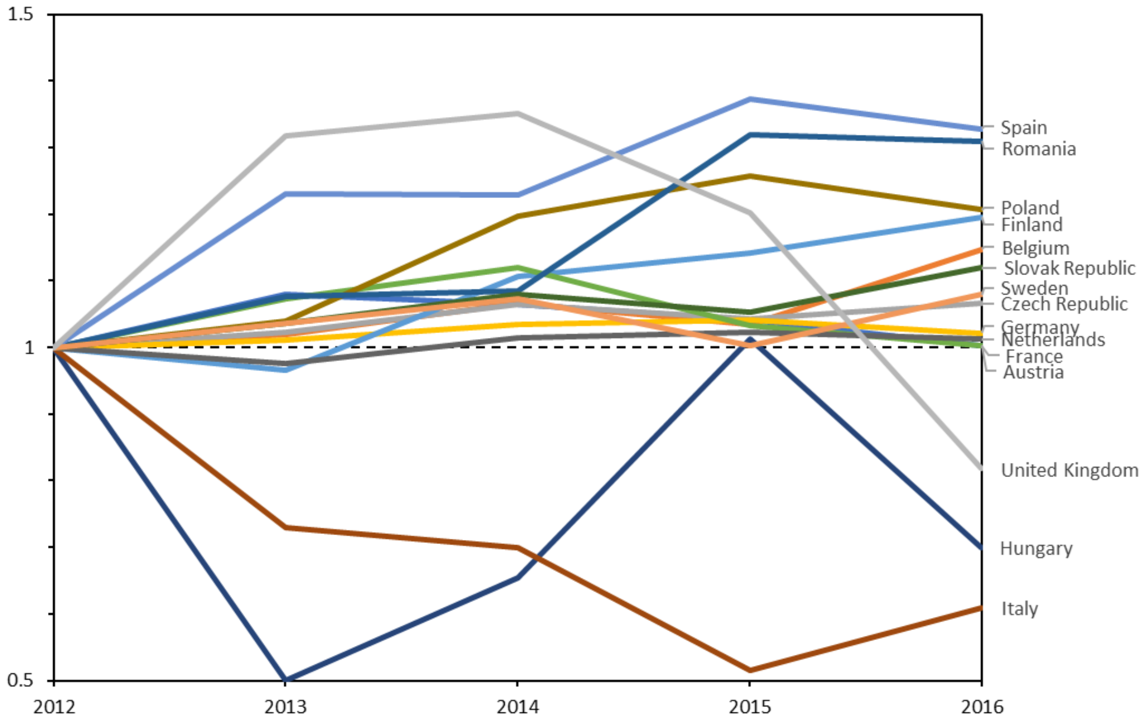
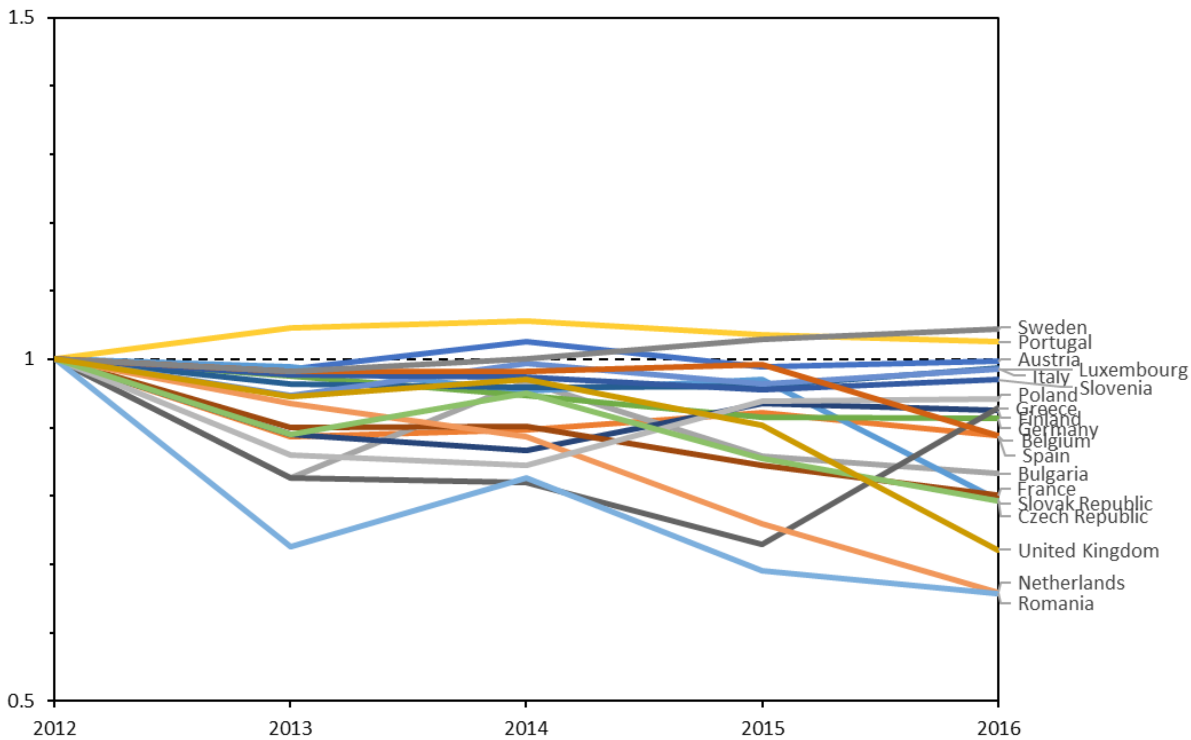


Table 2-8:Steel production from EAFs in the EU 2012-2016 (kt) (source: World Steel)

	2012	2013	2014	2015	2016
Austria	674	664	691	667	672
Belgium	2654	2355	2379	2448	2357
Bulgaria	633	523	612	543	527
Croatia	1	135	167	122	0
Czech Republic	371	367	354	360	295
Germany	13789	13459	13062	12622	12594
Finland	1460	1300	1265	1365	1350
France	6102	5491	5498	5159	4886
Greece	1247	1030	1022	910	1158
Hungary	54	139	178	168	233
Italy	17939	17295	17200	17227	17704
Latvia	805	198			
Luxembourg	2208	2090	2193	2127	2175
Netherlands	141	132	125	107	93
Poland	4132	3551	3492	3877	3891
Portugal	1960	2050	2070	2030	2010
Romania	1591	1155	1314	1100	1046
Slovakia	381	339	362	326	302
Slovenia	632	618	615	604	613
Spain	10216	10042	10042	10144	9069
Sweden	1443	1418	1443	1485	1506
United Kingdom	2054	1942	1995	1856	1482
EU	70487	66292	66039	65247	63962

Figure 2-4: Steel production from EAFs in the EU 2012-2016 (2012=1) (source: World Steel)



Secondary benefits

Some abatement techniques provide secondary benefits in reducing emissions of other pollutants (which may be covered by separate BAT numbers). The secondary emissions benefits have also been calculated in the same way as for primary emissions (using reported data if available; estimating if not). However, all costs of the technique have been counted against the primary pollutant abated (i.e. for the main BAT Number for that technique).

The principal example of this is bag filters installed for BAT 20 in sinter strands providing secondary benefits in also reducing SO_2 and dioxin emissions. The assumptions made in this example are:

- In case of installation of a bag filter, SO_2 emission reductions were based on the assumed achievement of the IED BAT-AEL associated with BAT 22 point IV of 350 mg/Nm^3 with the prevailing IPPC BAT-AEL of 500 mg/Nm^3 as the counterfactual. This value of 350 mg/Nm^3 is the lower end of the IED BAT-AEL range, and has been assumed based on the two processes that reported data: one reduced from 389mg/Nm^3 to 298mg/Nm^3, and another sinter strand reduced from 850mg/Nm^3 to 300mg/Nm^3 with the fitting of a bag filter with sorbent injection. AELs were combined with assumptions on operating hours and flow rates (based on the VDEh database) to calculate emission reductions.
- For dioxins the IED upper BAT-AEL of 0.1 ng I-TEQ/Nm^3 was assumed for the BAT scenario in case of implementation of a bag filter and emissions in the counterfactual estimated using a bag filter efficiency of 75% for removal of dioxins. The efficiency of 75% was derived from the difference in BAT-AELs for bag filter (0.1 ng/Nm^3) and ESPs (0.4 ng/Nm^3).
- For Hg emissions, the overall impact of the BATC is expected to be low, because in the BREF emission figures for both ESP and fabric filters are reported already to be below the BAT-AEL. Higher figures up to 75 µg/Nm^3 are referred in the BREF but these are due to high mercury content in domestic ores and are deemed to be relevant for a rather limited number of installations. As these installations have been equipped with BAT technology already before 2010, no additional costs and benefits were assumed for mercury.

2.2.2.9 Calculation of benefits

Emissions savings have been directly monetised using damage costs per tonne abated converted to a standard of 2015 prices. The source of the damage costs is EEA (2014) for SO_2, NO_X, PM_{10} and PCDD/F. These are Member State specific for SO_2, NO_X and PM_{10}, and for these pollutants are the central value of the reported low[18] and high[19] values. In other words, the mean of the value when using statistical life (VSL) and value of a life year (VOLY) (thus the uncertainty assessment used +/- 50%). The mercury damage costs are from previous work completed by Ricardo for the Commission (Ricardo, 2017) which concluded that the EEA (2014) mercury damage cost omitted mortality impacts which should be a dominant factor in the damage cost, and consequently used a higher alternative mercury damage cost drawn from Nedellec & Rabl (2016).

The utilised damage costs are shown in Table 2-9. The damage costs in this method for Finland and Sweden are particularly low, which might otherwise suggest that measures in these countries are not cost-effective. However, the damage costs used are national averages and may not present the situation in the local context of the installations in question. This may particularly affect BAT 33 which is relevant to Swedish installations, and to a lesser extent BAT 51 which is of relevance to a few Finnish and Swedish installations. The lower damage costs in the case of these two Member States could potentially be explained by a significant distance from the location of industrial installations and the larger agglomerations, resulting in reduced population exposure to pollution.

[18] Based on the VOLY (Value of Life Years lost) valuation approach.

[19] Based on the VSL (Value of Statistical Life) valuation approach.

Table 2-9: Damage costs used for benefits calculations (2015 price year)

Member State	PM_{10} (EUR/kg)	SO_2 (EUR/kg)	NO_x (EUR/kg)	Mercury (EUR/kg)	PCDD/F (EUR/g)
Austria	59	47	20	53,164	32,444
Belgium	89	54	10		
Bulgaria	41	16	10		
Croatia	34	25	15		
Czech Republic	61	29	15		
Denmark	25	27	7		
Finland	9	10	3		
France	51	37	12		
Germany	76	46	16		
Greece	30	9	3		
Hungary	61	28	17		
Ireland	21	26	8		
Italy	79	37	19		
Lithuania	25	24	8		
Netherlands	82	60	12		
Poland	62	27	11		
Portugal	33	12	4		
Romania	55	25	17		
Slovakia	49	24	15		
Slovenia	53	38	21		
Spain	40	17	4		
Sweden	12	12	5		
United Kingdom	59	34	8		

The damage costs for PM_{10} have been applied to the quantities of PM_{10} emissions derived from dust emissions using the assumptions in Table 2-10.[20] Information on the proportion of total dust that is size fraction PM_{10} has been gathered from EEA (2016) and the German UBA (2010). This information suggests that the proportion of dust that is PM_{10} varies by process (and would be expected to vary by technique types fitted). The assumptions selected for each process have been chosen to best match the BATs with highest impacts.

[20] The PM_{10} damage costs from the EEA are based on a constant $PM_{2.5}/PM_{10}$ ratio of 0.65. For the assessment of benefits using damage costs, ideally the ratio of $PM_{2.5}$ to total dust would be used rather than of PM_{10} to total dust. However, this has not been used because the key source of assumptions on the PM fractions in dust is German UBA (2010) which only includes the PM_{10} fractions. This could add an error in the results, but it is considered to be small.

Table 2-10: Proportion of steelworks dust that is PM_{10}.

Process	PM_{10} proportion of total suspended particles		
	Source: EEA (2016)	Source: German UBA (2010)	Value selected
Iron and steel production (overall)	60%	-	60% (for coke ovens)
Sinter strand	50%	78% (stationary) 35% (diffuse)	78% (main impacts are BAT 20)
Pelletisation (Unabated)	50%	No info	50%
Blast furnace	62.5%	69% (stationary) 44% (diffuse)	62.5% (approximate mid-value)
Basic Oxygen furnace	91.4%	81% (stationary) 38% (diffuse)	38% (main impacts are BAT 78)
EAF	80%	71% (stationary) 17% (diffuse)	71% (primarily bag filter impacts)

2.3 Findings

2.3.1 High-level estimate of the probable emissions from the installations prior to, and those to be expected after, implementation of the BATC

The results of three high level methods to estimate the impacts of the implementation of BAT conclusions for emissions to air are summarised in Figure 2-5. It is important to note that the three methods are significantly different - for example Methods 1 and 3 are at sector level, whereas Method 2 was based on process level analysis.

Method 1, which compares reported emissions to air with emissions projected from the year 2012 using production quantities, i.e. before the implementation of BAT conclusions, shows that in the case of many pollutants, reported emissions are lower than projections. This is particularly the case when looking at reported emissions in E-PRTR, where in 2016 there is a 27% reduction in SO_2, 13% reduction in NOx and a 25% reduction in emissions of Hg to air comparing reported emissions with those projected from 2012.

Using the outputs of Amec Foster Wheeler (2015) as Method 2, emissions reductions are generally higher. The IED scenario produced emission reductions compared with the baseline of -53% for NOx, -29% for SOx, although Hg reductions are low at only -0.7%.

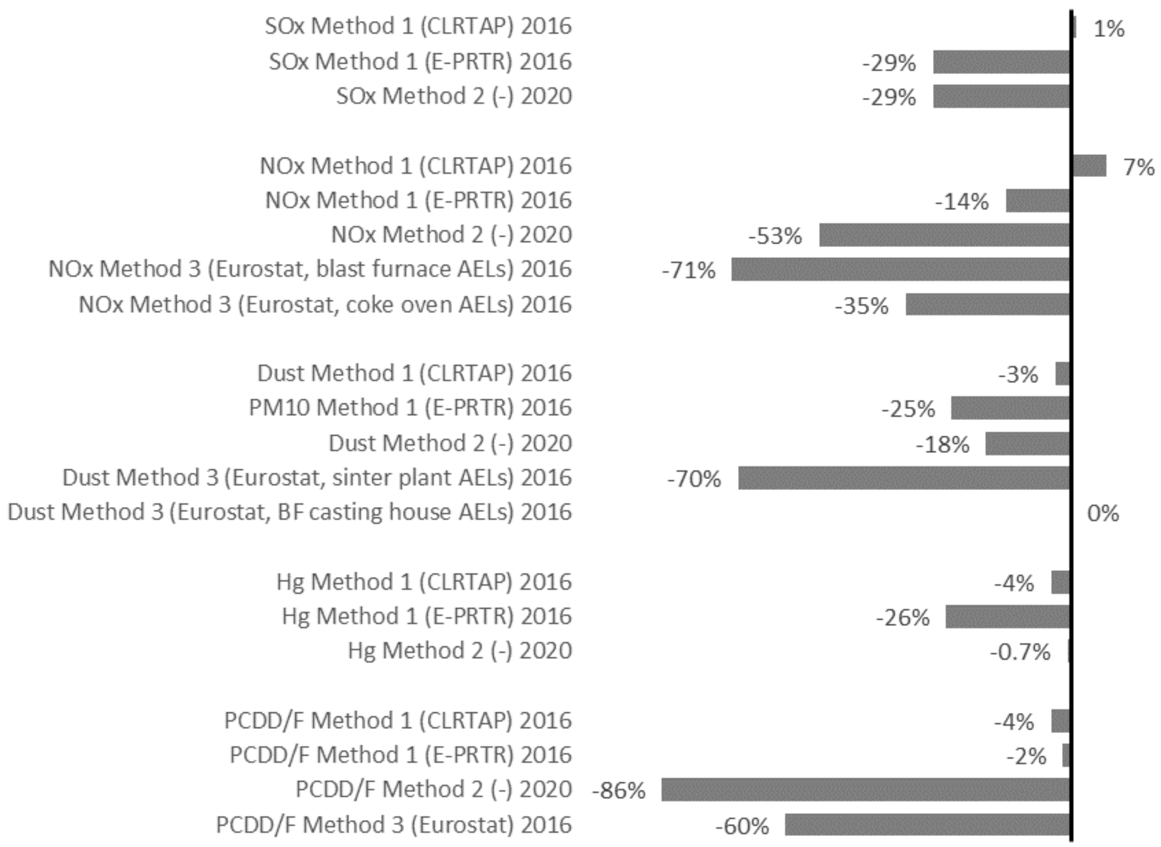
For Method 3, comparing IPPC BAT-AELs with IED BAT-AELs to estimate emission reductions, IED BAT-AELs are more stringent in most cases leading to anticipated emissions reductions. This includes 35% reductions in NOx emissions from coke ovens and 71% reductions from hot blast stoves, and 70% reductions in dust emissions from sinter plants. However, in some cases IPPC and IED BAT-AELs are identical, leading to no anticipated additional reductions in emissions. Method 3 omits some pollutants in the cases that IPPC BAT-AELs were not derived for the pollutants.

For emissions to water, it was deemed that the emissions data from E-PRTR represented too few facilities reported to enable robust analysis for method 1. Method 3 was not applied for emissions to water because for the most part there were no water pollutant IPPC AELs.

The findings from Figure 2-5 are:

- In almost all cases, all three high-level methods estimate reductions in emissions due to implementing the I&S BATC. However, this is not the case for every pollutant/method combination: an exception is for one sensitivity of Method 1 for both NO_X and SOx.
- The three high level methods have a very wide range in the predicted impacts on emissions for all air pollutants in the figure, leading to low confidence in predicted impacts.
- Between the methods, there isn't a consistent outcome that one Method estimates more extreme or more subtle changes; Method 2 from Amec Foster Wheeler (2015), which is arguably the most detailed method due to its assumptions made on dividing installation emissions among the constituent processes, for example predicts the highest PCDD/F impacts of all three methods, and the smallest Hg impacts of all three methods.
- For emissions to air of NO_X, dust (PM_{10} when observing E-PRTR), and PCDD/F, Method 2 and Method 3 (comparison of IPPC and IED BAT-AELs) predict larger reductions than are observed when comparing reported emissions with projected emissions (Method 1).

Figure 2-5: Comparison of high level methods (all pollutants and sensitivities) assessing BATC impacts on emissions to air, expressed as percentage change from a scenario without BATC



2.3.2 Summary of detailed assessment of costs and benefits of complying with the BATC

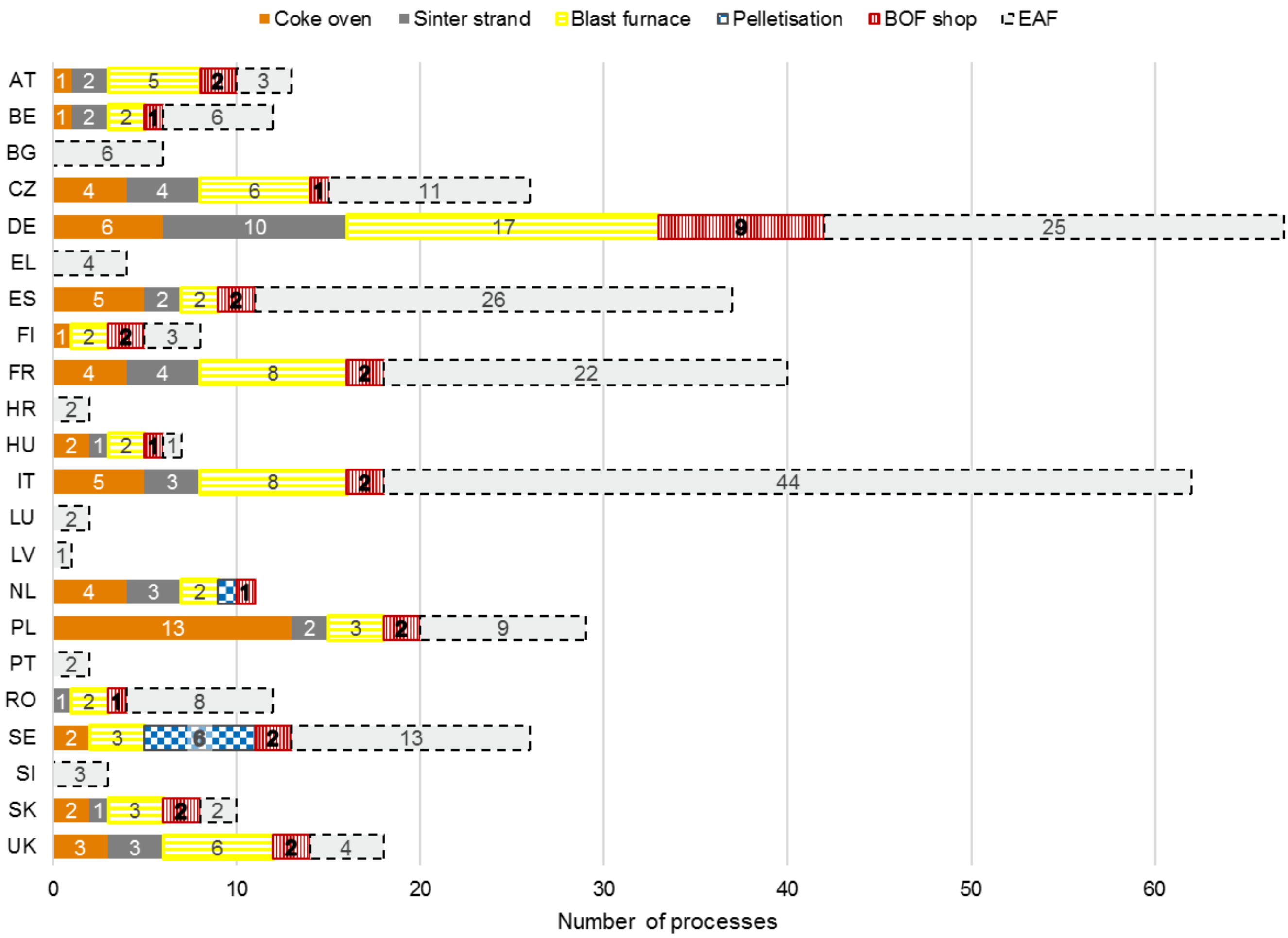
2.3.2.1 Installations and processes in assessment

The baseline database has overall, for the EU28, the following processes[21]:

- 53 coke oven plants (some as standalone coke oven installations; some as part of integrated steelworks)
- 38 sinter strands
- 71 blast furnaces[22]
- 7 pelletisation plants
- 32 BOF shops
- 197 EAFs (a small fraction of which are at integrated steelworks)

The number of processes split by type of process and by Member State, is shown in Figure 2-6.

Figure 2-6: Number of each process at each Member State included in the model (source: this study)



[21] Processes marked in the VDEh database as mothballed or shutdown were excluded from the baseline database. This included for example 5 EAFs marked as shutdown in the period 2012-2016.

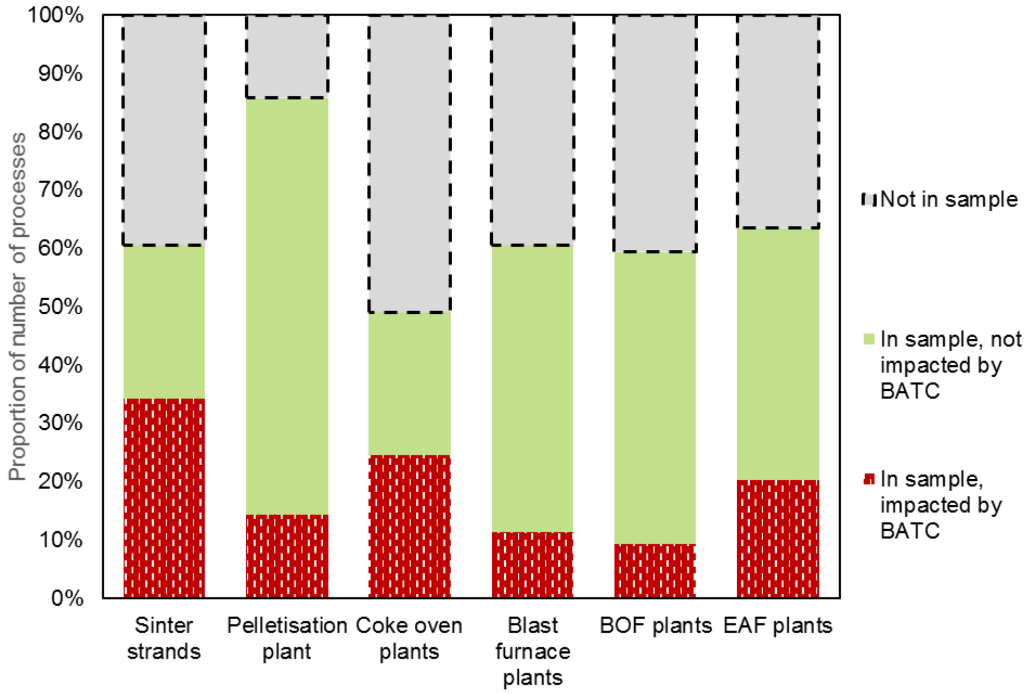
[22] This includes one blast furnace in Sweden which has been reported by the competent authority to be is not operational / in stand-by, and one unknown blast furnace in Germany which has been reported by industry as closed. However, these processes are not assumed in the analysis to be affected by the BATC so they do not appear in the results of costs or benefits.

Few integrated steelworks installations also have electric arc furnaces (only in Czech Republic and Finland), and few integrated steelworks installations have pelletisation plants (Sweden and the Netherlands). For the rest of the integrated steelworks, the proportion of each process type approximately scales with the overall number of installations, apart for Poland which has a higher proportion of coke ovens including 8 standalone coke oven installations. Overall, Germany has 20% of the integrated steelworks processes of the EU. Italy has the highest number of EAFs, with 22% of the EU total.

2.3.3 Summary of estimated costs and benefits of compliance with the BATC

For this assessment of the I&S BATC, information was sought on the impacts on all processes in the EU28. The information obtained represents, across the different processes, a sample size of around two thirds of total EU28 capacity (between 56% and 84%). *Within this sample*, the proportion of processes that have been identified as needing to fit or upgrade techniques to comply with the BATC is highest for sinter strands (57%), coke ovens (50%) and EAFs (32%). The impacts of the BATC on pelletisation plants, blast furnaces and BOFs appear to be lower, affecting around one sixth of these processes. This is summarised in Figure 2-7.

Figure 2-7: Proportion of processes impacted or not by the BATC in the sample, and proportion not included in the sample due to lack of available information. The processes that have been impacted the most by the I&S BATC are sinter strands, coke ovens and EAFs according to the sample assessed.



The summary of the estimated costs and benefits of fitting techniques to comply with the BATC is in Table 2-11 for the sample assessed. Techniques installed to reduce dust emissions are the most significant cost to operators, with the five highest cost BATs addressing dust emissions. BAT 20 (dust from sinter strands) leads to the highest costs on operators, with capital costs of €180 million, annualised to €13.2 million per annum.

In addition to BATs for dust emissions, some significant investments have been made to address other pollutants, with €12.9 million per year to reduce SO_2 from coke ovens, and €4.9 million per year to reduce SO_2 from sinter strands.

Table 2-11: Summary of costs and benefits.

Process	Number of processes impacted / in sample / in EU total[23]	Total annualised costs (€m/yr)	Benefits (€m/yr) of BATC compliance			
			Processes with reported emissions	Remaining processes (estimated emissions)	Total for all processes impacted	Benefit-cost ratio
Sinter strands	13 / 23 / 38	45.9	242	402	644	14.0
Pelletisation plants	1 / 6 / 7	7.7	6.8	-	6.8	0.9 (Note 1)
Coke ovens	13 / 26 / 53	17.1	154	1	155	9.0
Blast furnaces	8 / 43 / 71	4.7	3.3	12.5	15.8	3.3
BOF	3 / 19 / 32	12.2	45.5	-	45.5	3.7
EAFs	40 / 125 / 197	2.0	2.1	63.4	65.5	33
TOTAL	**-**	**89.6**	**453**	**479**	**932**	**10.4**

Note 1: The results for pelletisation plants reflect a single Swedish installation. The benefit-cost ratio here is thus particularly sensitive to the PM damage cost for Sweden, which is about one quarter of the EU average.

The monetised health and environmental benefits arising from the emissions reductions overall are greater than the estimated costs of the techniques for compliance. Overall the estimated costs representing approximately two thirds of the industry (based on the sample sizes discussed above) are €90m/yr (capital cost: €506m, and operating cost €52m/yr). Even after accounting for uplifting to represent to the whole EU28 industry, these values are lower than those anecdotally quoted by EUROFER in the course of the Ricardo (2016) study as €12bn. The anecdotal figure from EUROFER does not have any further supporting information as to its scope; it could be total impacts quoted as a net present value. In an approximate comparison, the annualised value in this study of €90m/yr for two thirds of the industry could mean that the total impact for the whole industry is ~€134m/yr, and over the assumed appraisal period of the technique lifetime of 20 years is €2.7bn.

The following section provides full detailed results of the assessment for each process and BAT number.

2.3.4 Detailed results of the impacts per process

2.3.4.1 Sinter strands

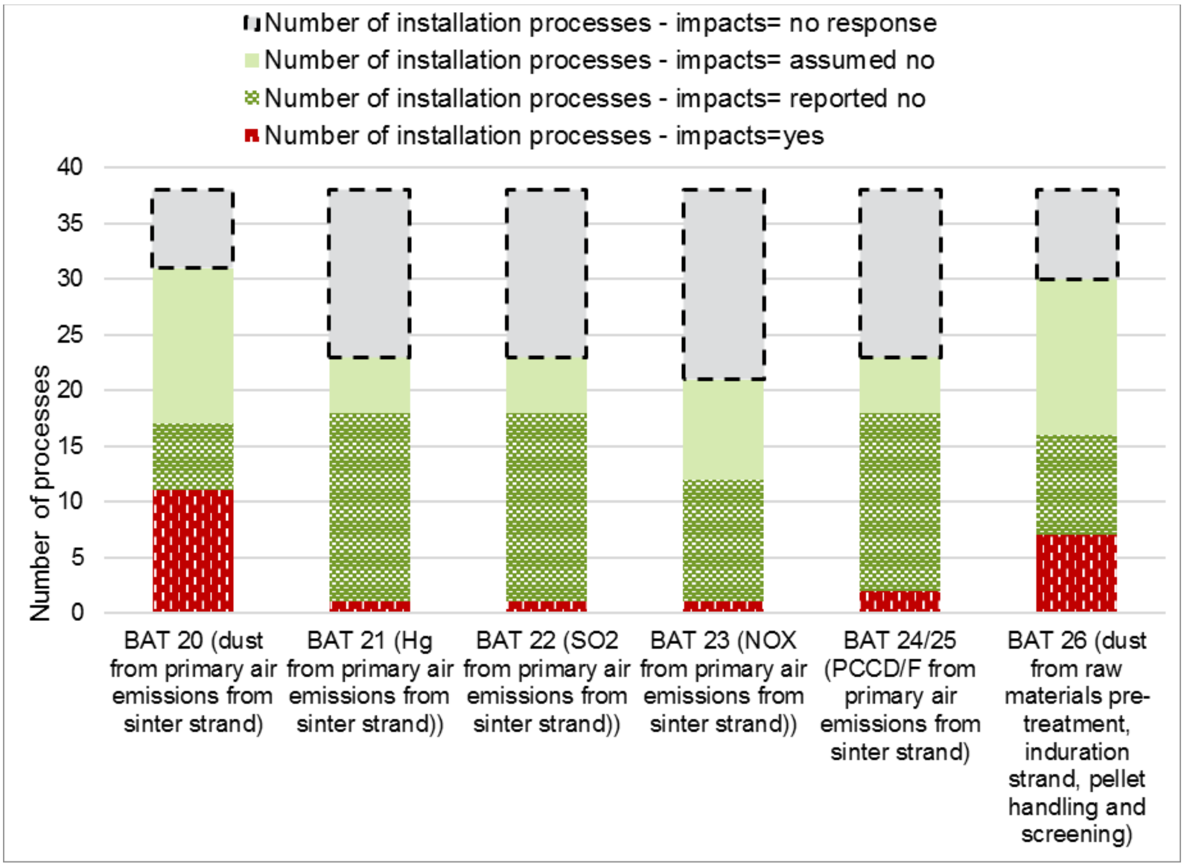
The BATs considered in the assessment were:

BAT 20 (dust from primary air emissions from sinter strand)
BAT 21 (Hg from primary air emissions from sinter strand)
BAT 22 (SO_2 from primary air emissions from sinter strand)
BAT 23 (NO_X from primary air emissions from sinter strand)
BAT 24/25 (PCCD/F from primary air emissions from sinter strand)
BAT 26 (dust from raw materials pre-treatment, induration strand, pellet handling and screening)

[23] The EU total refers to all processes, including such where no information was acquired. The remaining columns in the table reflect costs and benefits for the processes in the sample.

The proportion of the EU28 sinter strands for which information has been identified on whether they were impacted by the BAT conclusions is 61%, and these sinter strands represent 68% of total EU28 sinter strand capacity. Of 38 sinter strands in the database, there were 17 with reported information on impacts (13 with impacts, and four reporting no impacts) as well as a further six installations where no impacts were assumed on the basis of BAT being fitted prior to the publishing of the BAT Conclusions. This means that there are 15 sinter strands with no information on whether they were impacted by the BATC. This information is summarised in Figure 2-8. Five of the sinter strands identified were reportedly subject to a derogation under Article 15(4).

Figure 2-8: Information identified for sinter strand BATC impacts.



Note: most of the impacts counted under BAT 22 and BAT 24/25 as "reported no" predominantly refer to secondary benefits on SO_2 and PCCD/F emissions due to the techniques fitted to comply with BAT 20.

The BAT with the most impacts for sinter strands is BAT 20. Eleven sinter strands have been identified that have been impacted (Table 2-12). Benefits are shown for reported emission reductions as well as those estimated as described in Section 2.2.2.8. The most common technique installed for meeting BAT 20 is a bag filter with 7 sinter strands reported to have installed them and a further 3 assumed to have installed them on the basis of a current derogation and bag filters being the most commonly used technique. The bag filter also leads to co-benefits of other pollutants, which have also been calculated, and are included in the BAT relevant for that pollutant in Table 2-12. Of the responses to the questionnaire, one operator responded to indicate that the choice of fitting a bag filter was to meet the BATC requirements across all of BATs 20 (dust), 21 (Hg) and 24/25 (PCCD/F). For the remaining operators it is assumed that the reported choice to fit bag filters was due to BAT20.

Table 2-12: Summary of costs and benefits for BATs for sinter strands.

BAT	Number of processes impacted	Total annualised costs (€m/yr)	Benefits (€m/yr) of BATC compliance			Benefit-cost ratio
			Based on reported emissions	Based on estimated emissions	Total (Note 2)	
BAT 20 (dust)	11	31.8	29.7	95	125	
SO_2 (BAT 22) co-benefits of BAT 20	9	-	168	296	465	-
PCCD/F (BAT 24/25) co-benefits of BAT 20	9	-	-	0.5	0.5	-
Subtotal BAT20		*31.8*	*198*	*392*	*590*	*18.6*
BAT 22 (SO_2)	1	4.9	10.6	-	10.6	2.2
BAT 23 (NO_x)	1	3.6	10.1	0	10.1	2.8
BAT 25 (PCCD/F) (Note 1)	2	2.8	-	0.3	0.3	0.1
BAT 26 (dust)	7	2.7	23.1	9.2	32.3	11.9
Total[24]	**13**	**45.9**	**242**	**402**	**644**	**14.0**

Note 1: the two sinter strands are reported to have implemented lignite injection as a technique to reduce PCDD/F to meet BAT25.

Note 2: for this table and others in the report, the total benefits is the sum of the total for installations with reported data on impacts and those installations with estimated benefits.

The impacts identified but not quantified for sinter strands are:

Table 2-13: Impacts not quantified for sinter strands

BAT	Impact	Aspect not quantified
BAT 20	2 sinter strands- Improvement to ESP	Capex and opex due to lack of cost data. Benefits were also not calculated as no information was found for the iron and steel sector. However, an indicative abatement efficiency of 40% was found for ESP upgrades in the large combustion sector. Due to the lack of cost data however this was not modelled.
BAT26	1 sinter strand - reducing secondary emissions from emptying and sorting the sinter belt	Capex, opex, emissions impacts due to lack of sufficient detail to estimate impacts
BAT26	1 sinter strand bag filter	Emissions impacts due to lack of flow rate data

[24] The total number of processes affected by the BATC are counted only at installation level. One installation process may be reported to be affected by the BATC for multiple BATs, but in such a case would only be counted once. The totals of costs and benefits are sums of the figures presented per BAT. Total annualised costs include both annualised CAPEX and annualised OPEX. The benefit-cost ratio was calculated dividing the total benefits by the total costs (annualised). Rounding is applied for presentational purposes but not in the calculations. The calculation method applies to all cost/benefit summary tables throughout the report.

BAT 20 – Dust from primary air emissions from sinter strands

BAT 20 is the most significant BAT in terms of the number of installation processes with reported impacts, as well as the magnitude of costs and benefits, with estimated initial investment costs of €180 million, and expressed as a total annualised cost with operating costs as €32 million/year and benefits from reduction in PM_{10} emissions valued at €125 million/year depending on estimate method used.

Most of these impacts are associated with the reported installation of bag filters to meet BAT AELs. Bag filters were reported to be installed by six sinter strands in the EU including at one installation which also implemented waste gas recirculation. Bag filters were assumed to be installed in three other installations due to derogations in BAT 20. Two sinter strands are reported to have made improvements to their ESP along with lignite injection, but these impacts have not been quantified (for costs or benefits) due to lack of additional detail. There were reported to be no impacts associated with BAT 20 in 6 installations. There were assumed to be no impacts in thirteen installations based on information in the VDEh database indicating that a bag filter (i.e. BAT) was already installed in 2010 or earlier. Due to the installation of bag filters, secondary benefits are estimated for SO_2 and dioxins. The SO_2 benefits overall are dominant, which appears to be principally driven by the damage cost for SO_2. The largest SO_2 emission reductions are based on reported concentration data, and so would appear to be reliable.

This information is summarised in Table 2-14. The BREF costs used to quantify impacts appear to include sorbent injection.

Table 2-14: BAT 20 impacts- Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr unless specified)	Benefits (€m/yr)
9 (no derogation)	7	141	10.4	13.0	PM_{10} (primary)	1,415 (range 1,122-1,707)	106 (range 87-126)
2 (with derogation)	2	39	2.9	5.6	PM_{10} (primary)	342 (range: 244-440)	18.5 (range 14.9 - 22.1)
(9)	*(9)*	*(Inc. above)*	*(Inc. above)*	*(Inc. above)*	SO_2 (secondary benefit)	9,143	465
(9)	*(9)*	*(Inc. above)*	*(Inc. above)*	*(Inc. above)*	PCDD/F (secondary benefit)	16.6 g Teq	0.54
TOTAL: 11	**9**	**180**	**13.2**	**18.6**			**124.9 excl. Secondary benefits**

Note: The information provided in the table on emissions and benefits is shown as a range, which represents the alternative values when using the two described methods for estimating emissions impacts.

BAT 21 – Hg from primary air emissions from sinter strands

One installation operator reported that the investment of a bag filter for a sinter strand was related to BAT 20, BAT21 as well as for BAT 24/25. However, because this investment was reported to occur in 2008 and cited other driver this was considered to be prior to the publication of the BATC and thus not directly attributable to the BATC. The costs and benefits for this were therefore not included here or under BAT 20.

For all the sinter strands for which bag filters were reported or assumed to be fitted to meet BAT20 have not had any co-benefits on mercury quantified. This is because it is understood that only a limited number of Member States use ores with high mercury content, and all other Member States use ores with low levels of mercury. There is an absence of information identified, including in the BREF, as to the variation in these remaining Member States in the Hg emission levels. In any case, the mercury co-benefits would be expected to be small due to the already low levels of mercury.

There are no additional impacts to report under BAT 21.

BAT 22 – SO_2 from primary air emissions from sinter strands

The SO_2 emissions impacts of all the bag filters reported or assumed to be fitted to meet BAT20 have been included under BAT20.

In addition, one installation is reported to have made an investment in sodium bicarbonate injection in 2011 to comply with the then-anticipated BATC to reduce SO_2 emissions, with a reported initial investment cost of €1.5 million and an additional annual operating cost of €4.8 million/year. This impact has been included as having occurred to meet the BATC because although it occurred prior to the final publication of the BATC, it is understood that the BAT-AELs were known to have been agreed by this time. The impacts of this are included in Table 2-15.

The SO_2 benefits from this reported instance of bicarbonate injection are small in comparison to the benefits provided by bag filters installed primarily to meet BAT 20.

Table 2-15: BAT 22 impacts- Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
1	1	1.5	0.1	4.8	SO_2	226	10.6

BAT 23 – NOx from primary air emissions from sinter strands

One installation is reported to have installed Selective Catalytic Reduction (SCR) to reduce NOx to meet BAT23 in 2012. This impact has been included as having occurred to meet the BATC because although it occurred around the final publication of the BATC, it is understood that the BAT-AELs were already known by the time of the investment. The impacts of this are included in Table 2-16.

Table 2-16: BAT 23 impacts- Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
1	1	15.8	1.2	2.5	NO_x	507	10.1

BAT 24/25 – PCDD/F from primary air emissions from sinter strands

In addition to the impacts on dioxins associated with bag filters installed to meet BAT 20 (impacts included under BAT20), one installation with two sinter strands is reported to have begun lignite injection from 2016. The impacts are summarised in Table 2-17. The dioxins benefits are small in comparison to the benefits provided by bag filters installed primarily to meet BAT 20.

Table 2-17: BAT 24/25 impacts- Costs and emission benefits

No. sinter strands with reported impacts	No. sinter strands with reported impacts quantified	Capital costs (€m)	Operating costs (€m/yr)	Pollutant	Emissions reductions (g Teq/yr)	Benefits (€m/yr)
2	2	-	2.8	PCDD/F	9.0	0.29

Note: The only reported impacts for BAT 24/25 direct impacts are lignite injection, which has no capital costs.

BAT 26 – Dust from secondary air emissions from sinter cooling and other relevant sources (discharge, crushing, screening, conveying)

In the case of one installation, a bag filter installed to reduce secondary dust emissions was reported with capital costs of €7 million and annual operating costs of €30,000, and a reduction of PM_{10} emissions of 32 tonnes/year which has been monetised as €1.9 million per year.

One installation reported installation of a second ESP to double the capacity of secondary dedusting and reported the emissions savings associated with this. Benefits were calculated reported emissions savings.

For one sinter strand, a bag filter was assumed to be installed following a derogation. The costs of this were extrapolated from costs reported from another sinter strand, scaled according to capacity. For this sinter strand no emissions benefit was estimated, due to a lack of data on flow rate in the VDEh database for this installation.

Three sinter strands, one of which applied for a six month derogation to delay the investment to autumn 2016, report needing to upgrade ESPs. These impacts have been costed using data from the BREF as none were available in the permit decision document. The permit decision document for one of the sinter strands provided mass emissions and emissions concentration data; the emissions concentration data have been used to extrapolate the emissions impacts for the remaining two sinter strands.

One sinter strand reported that they reduced secondary emissions from emptying and sorting the sinter belt. However, no costs nor benefits are identified for this impact.

The impacts for BAT26 are summarised in Table 2-18.

Table 2-18: BAT 26 impacts - Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
7	6 (costs), 5 (benefits)	33.7	2.5	0.24	PM_{10}	504	32.3

Note: One benefit was not monetised due to lack of adequate data in the reported technique (Reducing secondary emissions from emptying and sorting the sinter belt). For further details on impacts that were not quantified see Table 2-13.

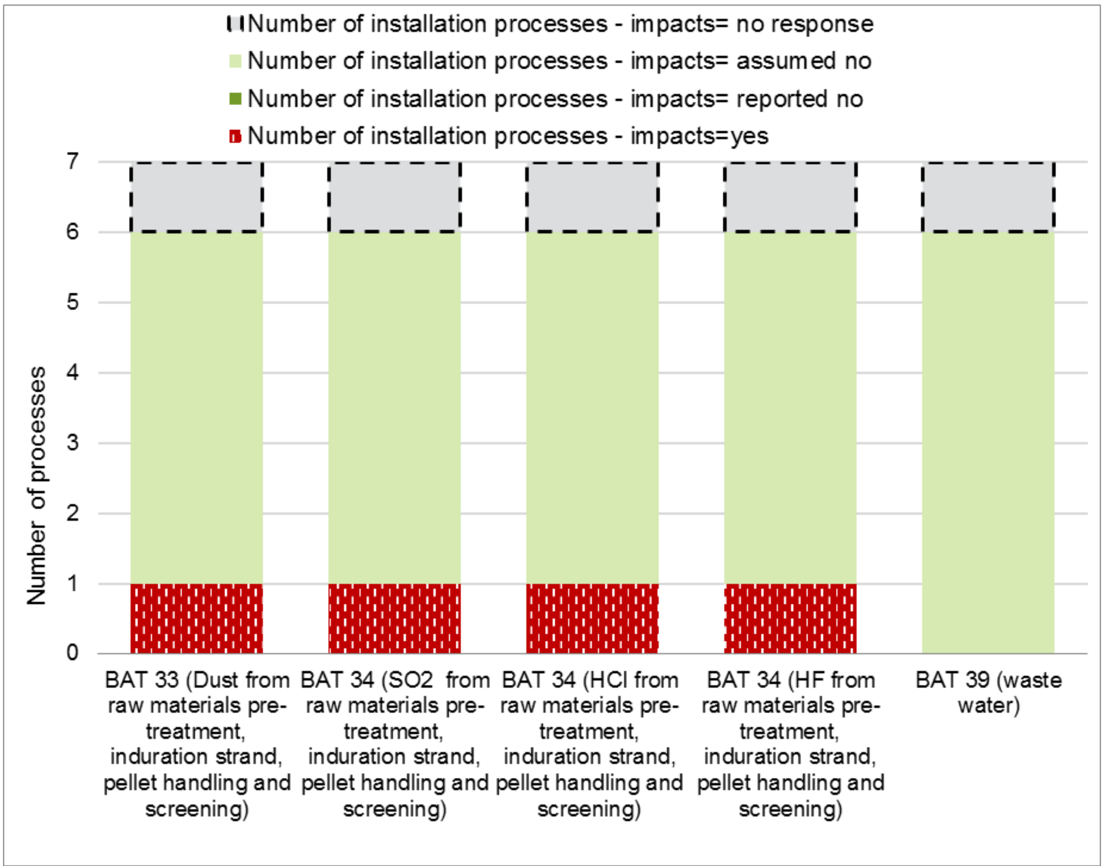
2.3.4.2 Pelletisation plants

The BATs considered in the assessment were:

BAT 33 (dust from raw materials pre-treatment, induration strand, pellet handling and screening))
BAT 34 (SO_2, HCl and HF from raw materials pre-treatment, induration strand, pellet handling and screening)

Only 7 pelletisation processes have been identified as operating across the EU28. Six of them are located at four installations. **Information has been identified for 6 pelletisation processes on whether they were impacted by the BAT conclusions (~86%), and these pelletisation plants represent 84% of total EU28 pelletisation plant capacity.** Information for the pelletisation plants has been identified predominantly from discussions with a national competent authority and from a review of the permits of the plants. This information is summarised in Figure 2-9 and Table 2-19.

Figure 2-9: Information identified for pelletisation BATC impacts.



The results for pelletisation plants in Table 2-19 reflect a single installation in Sweden. As such, the benefit-cost ratio results for pelletisation plants are particularly sensitive to the damage cost for Sweden. The PM damage costs used for Sweden are about one quarter of the EU average PM damage costs. Consideration of the site-specific situation could lead to re-evaluation of the appropriateness of applying the Swedish average damage cost.

Table 2-19: Summary of costs and benefits for BATs for pelletisation plants.

BAT	Number of processes impacted	Total annualised costs (€m/yr)	Benefits (€m/yr) of BATC compliance			Benefit-cost ratio
			Based on reported emissions	Based on estimated emissions	Total	
BAT 33	1	7.7	6.8	-	6.8	0.9
BAT 34	1	-	-	-	-	-
BAT 39	0	-	-	-	-	-
Total	**1**	**7.7**	**6.8**	**-**	**6.8**	**0.9**

Summary of impacts identified but not quantified for pelletisation plants:

BAT	Impact	Aspect not quantified
BAT 34	1 pelletisation plant's new flue gas treatment	Capex, Opex, emissions impacts, benefits – Due to lack of sufficient information on the nature of the technique to estimate impacts

BAT 33 – Dust from raw material pre-treatment, induration strand, pellet handling and screening

One pelletisation process has been identified as having incurred impacts to comply with both BAT 33 and BAT 34. For compliance with BAT33 this process fitted a new bag filter in 2013; this impact has been quantified using reported data from the permit to estimate the benefits and using cost data from the BREF to estimate the plant costs. The pelletisation process in question is not subject to a derogation under Article 15(4).

The impacts for BAT 33 are summarised in Table 2-20.

Table 2-20: BAT 33 impacts - Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
1	1	19.6	1.4	6.3	PM_{10}	561	6.8

BAT 34 – HF, HCl and SO_2 from raw material pre-treatment, induration strand, pellet handling and screening

One pelletisation process has been identified as having incurred impacts to comply with BAT 34. For compliance with BAT34, this process fitted new flue gas treatment in 2015, for which the impacts (neither costs nor benefits) have not been calculated.

BAT 39 – Suspended solids, Kjeldahl nitrogen, COD and heavy metal emissions in waste water

Inspection of environmental reports (i.e. for 6 of 7 pelletisation plants) confirmed that no impacts arose at these installations for BAT39.

2.3.4.3 Coke ovens

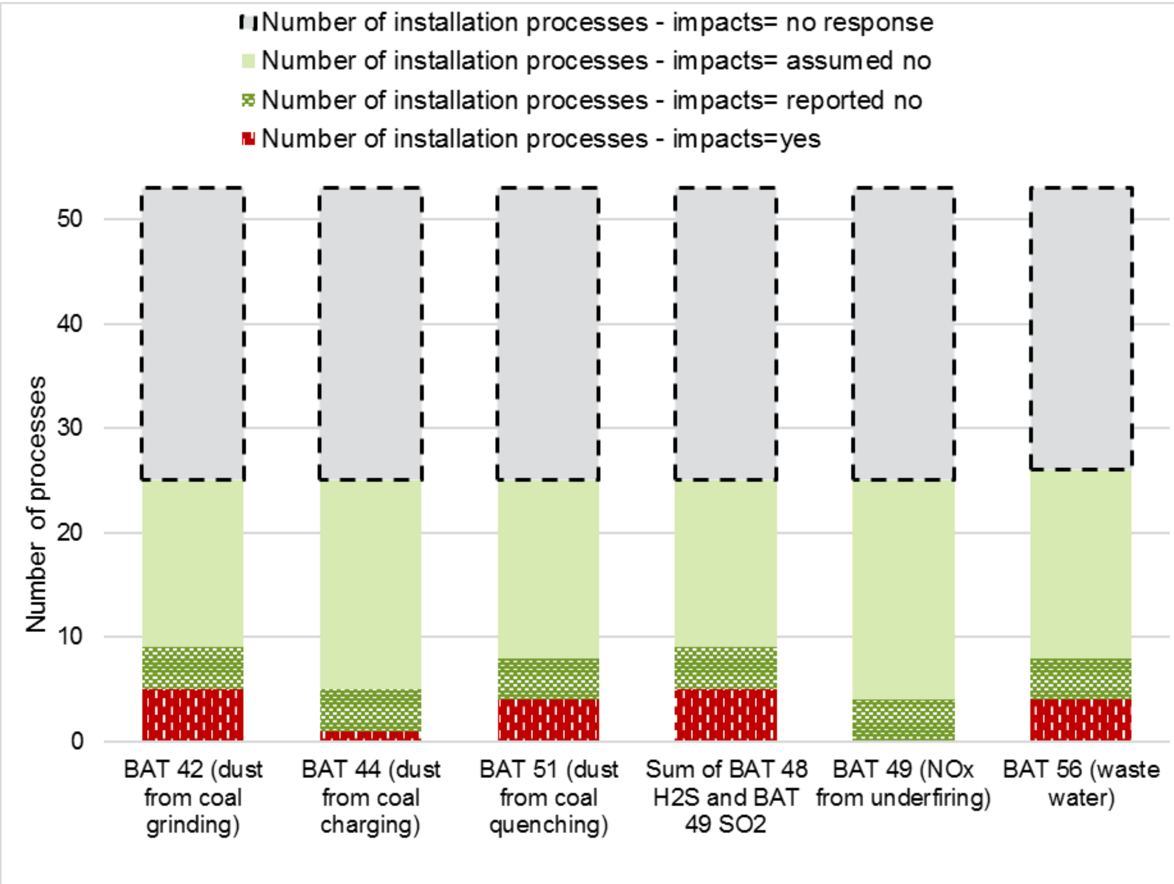
The BATs considered in the assessment were:

BAT 42 (dust from coal grinding)
BAT 44 (dust from coal charging)
BAT 51 (dust from coke quenching)
BAT 48 (H_2S from COG desulphurisation)
BAT 49 (SO_2 from underfiring)
BAT 49 (NO_X from underfiring)
BAT 56 (waste water)

The proportion of the EU28 coke ovens for which information has been identified on whether they were impacted by the BAT conclusions is 49%, and these coke ovens represent 56% of total EU28 coke oven capacity. Of 53 coke oven plants in the database, there were 18 with reported information on impacts (13 with impacts, and five reporting no impacts). There were also a further eight installations where no impacts were assumed based on having received responses from Member State authorities on the impacts of the BATC on integrated steelworks that did not mention any impacts for coke ovens. Overall, this means that for just over half of all coke ovens no information was identified on whether they were impacted by the I&S BATC. This is a significant gap, with the majority of these installations in countries where no information was gathered from the Member State proformas. This information is summarised in Figure 2-10 by BAT number. Eight of the coke ovens identified were reportedly subject to a derogation under Article 15(4).

The main BATs affecting operators of coke ovens appear to have been BAT 51, and BAT 48/49. BATs 48 and 49 have been grouped because both affect SO_2 emissions. With regard to BAT 49, the impact has been calculated under the assumption that H_2S in the coke oven gas has been converted to SO_2 when burning coke oven gas in other units (e.g. power plants or ferrous metal processing plants).

Figure 2-10: Information identified for coke ovens BATC impacts.



Note: most of the impacts counted under sum of BAT 48 and 49 are attributed to BAT 48.

The overall impacts are shown in Table 2-21. Several of the BATs are shown to have benefit-cost ratios below 1.

Table 2-21: Summary of costs and benefits for BATs for coke ovens.

BAT	Number of processes impacted	Total annualised costs (€m/yr)	Benefits (€m/yr) of BATC compliance			Benefit-cost ratio
			Based on reported emissions	Based on estimated emissions	Total (Note)	
BAT 42 (dust)	5	0.6	0.1	0.3	0.4	0.6
BAT 44 (dust)	1	0.2	0.1	-	0.1	0.5
BAT 51 (dust)	4	3.5	2.5	-	2.5	0.7
BAT 48 (H_2S) / BAT 49 (SO_2)	5	12.9	151	0.6	152	11.8
BAT 49 (NO_X)	0	-	-	-	-	-
BAT 56 (waste water)	4	-	-	-	-	-
Total	**13**	**17.1**	**154**	**1**	**155**	**9.0**

Note: benefits shown are the mid-point values of ranges in some cases. Those ranges are shown in the detailed tables below.

Summary of impacts identified but not quantified for coke ovens:

Table 2-22: Impacts not quantified for coke ovens

BAT	Impact	Aspect not quantified
BAT 51	4 coke oven plants Capex only quantified.	Opex- No costs mentioned from two sources- Assumed to be no operating costs
BAT 49	1 coke oven plant with 2 year derogation	Capex, Opex, emissions impacts
BAT 56	3 coke oven plants with derogations	Capex, Opex, emission impacts- No information on what technique(s) will be installed

BAT 42 – Dust from coal grinding

Five coke oven plants have been identified as having to make changes to comply with BAT 42. This has been represented as follows:

- One coke oven has implemented efficient extraction and subsequent dedusting, and has reported data on the costs and emissions impacts of this, with emission savings of 2.4 tonnes of dust (saving of 1.4 t PM_{10}) per annum valued at €86,000/year.
- Four coke oven plants (across three installations) have been identified, according to the received operator questionnaire and Member State proforma, as needing to fit dry dedusting equipment to replace existing wet dedusting equipment, at a capital investment of €7.4m. Of these, there is one non-integrated coke oven installation which has reported costs and achieved dust emission concentration for this technique. The information for this installation appears to indicate that the existing techniques were relatively old and would have needed replacing anyway under business as usual. These impacts have been counted, but there is an argument to assume such costs and impacts would have been borne under BAU (counterfactual). For the remaining coke ovens, the costs of the bag filters have been extrapolated on a capacity basis from the installation which provided the cost data. Assuming a post-abatement concentration of 0.75mg/Nm^3 in all four coke ovens (based on reported concentrations at one of the four coke ovens), benefits are small when estimating the counterfactual using abatement efficiency (99%

post-BAT, 95% pre-BAT). However when estimating using an IPPC concentration of 50mg/Nm^3 benefits are greater. The difference between these methods is reflected in the large range of estimated benefits (Table 2-23).

The calculated costs and benefits are summarised in Table 2-23.

Table 2-23: BAT 42 impacts - Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
5	5	7.5	0.55	0.01	PM_{10}	6.0 (range 2.0-10.1)	0.4 (range 0.1-0.6)

BAT 44 – Dust from coal charging

Only one operator has reported an impact from needing to make changes to the coke ovens to comply with BAT 44. The operator has reported costs and emissions impacts for implementing efficient extraction and subsequent dedusting in the charging car in 2014. These impacts are summarised in Table 2-24. However, with regard to BAT44 not only are dust emissions relevant, but also organic compounds. Due to the monitoring methods (visible emissions) and due to the lack of damage costs for this pollutant, no benefits for organic compounds could be calculated. However, taking into account additional benefits from the abatement of organic compounds, the overall benefits might be higher than indicated in the table.

Table 2-24: BAT 44 impacts- Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
1	1	2.4	0.2	0.04	PM_{10}	1.8	0.11

BAT 51 – Dust from coke quenching

Four coke oven processes are reported to be impacted by BAT51. Of these, two had/have Article 15(4) derogations. The calculations carried out are as follows:

- In one installation, according to the environmental report of the installation, there was a new quenching tower reported to have been installed with reported benefits of 44 tonnes dust/year but costs were not reported. Costs were instead estimated from the value of a quenching tower reported as incurred at another installation, scaled by capacity.
- In one installation a new quenching tower was also reported, with reported costs available. The benefits have been estimated based on the difference between the BREF-based emission levels of existing quenching towers of 50g/tonne coke, and the BAT-AEL of 25 g/tonne coke.
- In one installation (2 coke oven processes), coke quenching abatement was reported to be installed by 2024 with estimated costs and benefits calculated as part of the derogation process within the environmental permit.

Table 2-25: BAT 51 impacts- Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t /yr)	Benefits (€m/yr)
2 (no derogation)	2	34.8	2.56	0	PM_{10}	45	1.7
2 (with derogation)	2	12.3	0.91	0	PM_{10}	11.8	0.7
Total	**4**	**47.1**	**3.5**	**0**		**57**	**2.5**

The results for BAT 51 include one installation in Sweden for which the PM damage costs are about one quarter of the EU average PM damage costs. The benefit-cost ratio for BAT 51 would increase from 0.7 to around 1.0 if the average EU damage costs were used for the Swedish process.

BAT 48 – H_2S from COG desulphurisation and BAT 49 – SO_2 from underfiring

BATs 48 and 49 (for SO_2) have been considered in combination as both impact on SO_2 emissions.

One coke oven plant operator reported an upgrade of wet oxidative desulphurisation was implemented to comply with the I&S BATC, with a reduction in H_2S concentrations from 50 to 10 mg/Nm^3. From this, SO_2 benefits were estimated at 19t per annum, monetised at €566,000/year. Costs were estimated from the BREF based on operational costs of flue gas desulphurisation. A second installation has reported impacts of BAT49, but no further information is available; this impact has not been quantified.

The environmental permit decision document of one installation with two coke ovens includes a cost-benefit analysis supporting a derogation for BAT 48/49. In this the estimated costs and benefits of coke oven gas desulphurisation are described, with capital costs of €34 million, and with emission savings of 2,378 tonnes/year, which when valued with the EEA damage costs is €80.6 million/year. This leads to a benefit-cost ratio when considering annualised costs of 16.8 for this installation.

A further installation is also identified with a derogation for BAT 48/49. Again based on a permit decision document, this installation is assumed to incur impacts from installing coke oven gas desulphurisation, with capital costs of €27 million and annual SO_2 emission savings valued using EEA damage costs at €70.8 million.

It is noted that the technique of coke oven gas desulphurisation can lead to high costs and high benefits. If further information on implementation of this technique to comply with the BATC then the conclusions and benefit-cost ratio could change significantly given that coke ovens had the smallest proportional sample (50%) of total EU industry of all the processes assessed.

Table 2-26: BAT 48 (H_2S) and BAT 49 (SO_2) impacts- Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
1 (no derogation)	1	-		0.58	SO_2	19	0.57
5 (with derogation)	3	61.1	4.5	7.8	SO_2	4,467	151
Total	**4**	**61.1**	**4.5**	**8.4**		**4,486**	**152**

BAT 49 – NO_X from underfiring

No identified impacts for BAT 49, although see “Other BATs” below.

BAT 56 – Emissions in waste water

Four coke oven plants have been identified as having been impacted when needing to comply with BAT 56. All four are identified as having obtained a derogation under Article 15(4). However, insufficient information was available to quantify the impacts.

Other BATs

Some information was provided on impacts which were deprioritised at the scoping stage. They are not included in the core CBA but are presented below for illustrative purposes:

For one Member State, assessment of a permit indicates that one coke oven plant incurred an impact associated with the dust BAT-AEL in BAT 49. That part of BAT 49 was deprioritised from scope of this assessment (see section 2.1.3). Another Member State has indicated that meeting the dust BAT-AEL for coke oven underfiring in BAT 49 is a challenge for many older or poorly maintained coke ovens, because abatement techniques of e.g. bag filters are not applicable as they would hinder the natural draft of the stacks. However, it is considered that, since no further information on this BAT 49 for dust was identified in the study, that it is not necessarily a wider issue common to more installations. It is nevertheless acknowledged that the upper BAT-AEL of 20 mg/Nm^3 might be difficult to meet in older coke ovens if there is cross wall leakage. From the BREF referring to repair measures and from the split view recorded in the BREF on this point (BREF page 565) it appears that older plants have been taken into account when deriving BAT, suggesting the BAT-AEL is achievable by older plants.

One Member State reported that a derogation had been granted for one integrated steelworks for BAT 50, which is for dust from coke pushing. BAT 50 was deprioritised from scope of this assessment. The Member State authority did not provide the end date, and indicates that an ELV of 30mg/Nm^3 was granted rather than the BAT-AEL of 10mg/Nm^3. No other information on this BAT was provided.

2.3.4.4 Blast furnaces

The BATs considered in the assessment were:

BAT 59 (Dust from storage bunker of coal injection)
BAT 61 (Dust from cast house)
BAT 65 (Dust from hot stoves))
BAT 65 (SOx from hot stoves)
BAT 65 (NOx from hot stoves)
BAT 67 (Suspended solids, cyanide, iron, lead, zinc in waste water from blast furnace gas treatment)

The proportion of blast furnaces with information on BAT conclusion impacts is 61% of all blast furnaces in EU28, or 57% of the total EU28 capacity. Expressed in numbers, information was identified for 43 out of 71 blast furnace installations. Only 8 of these have been identified as being impacted by the BAT conclusions and 10 have reported no impacts. No impacts were assumed for an additional 25 installations, 17 of which reported that there were either low or non-existent impacts and the remaining 8 did not discuss impacts on blast furnaces in their response to the questionnaire. No blast furnaces were identified as subject to a derogation under Article 15(4). No response was received from the remaining 28 blast furnace installations. A summary of the results is presented in Figure 2-11.

Figure 2-11: Information identified for blast furnace BATC impacts.

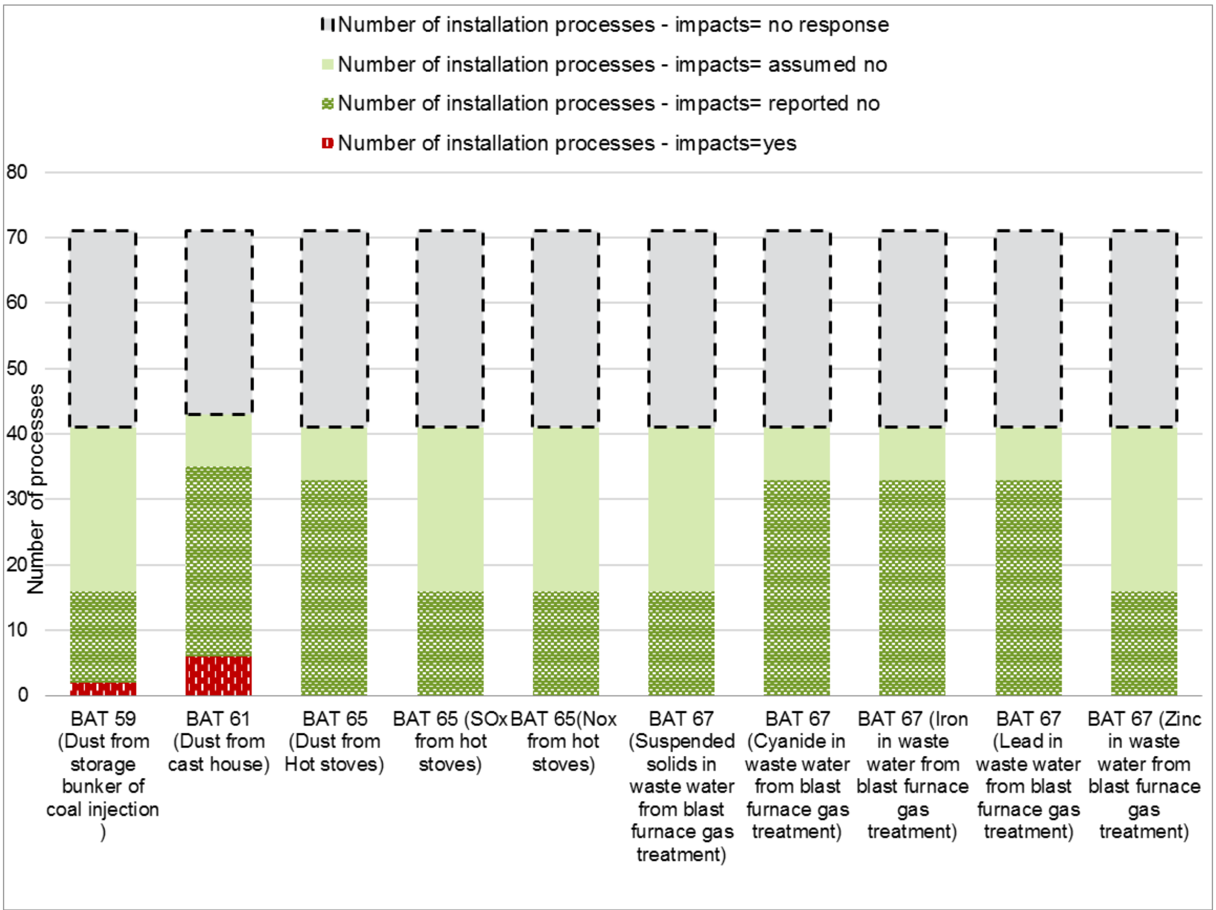


Table 2-27: Summary of costs and benefits for BATs for blast furnaces.

BAT	Number of processes impacted	Total annualised costs (€m/yr)	Benefits (€m/yr) of BATC compliance			Benefit-cost ratio
			Based on reported emissions	Based on estimated emissions	Total	
BAT 59	2	0.1	0.02	-	0.02	0.2
BAT 61	6	4.6	3.3	12.5	15.8	3.4
Total	**8**	**4.7**	**3.3**	**12.5**	**15.8**	**3.3**

BAT 59 – Dust from storage bunker of coal injection

The two blast furnaces that reported impacts under BAT 59 are in the same installation. The same technique was reported as installed in 2013 in both blast furnaces and involved the optimisation of the capture efficiency for diffuse dust emissions and fumes, and the subsequent off-gas cleaning by means of an electrostatic precipitator or bag filter. Costs and benefits shown here were based on reported information by the operator. It is unclear whether these impacts would have been borne under business as usual as the plants did not have coal injection for the blast furnaces previously. The combined investment costs are reported as €817,000 compared to low reported benefits of a PM_{10} reduction of 324 kg per year, valued at €19,000/year. A summary of the findings is presented in Table 2-28.

Table 2-28: BAT 59 - cost and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
2	2	0.8	0.06	0.05	PM_{10}	0.3	0.02

BAT 61 – Dust from cast house

There are more significant costs and benefits associated with BAT 61 than BAT 59. A summary of the findings is presented in Table 2-29.

Cost impacts were quantified with reported data for one out of six blast furnaces and estimated for the other five. One blast furnace reported implementing an upgrade of a dry dedusting system with associated operational costs (only) of €0.14m/yr. Emission impacts were quantified with reported data for four out of the six blast furnaces and estimated for the remaining two. Emission benefits for two blast furnaces were estimated based on an assumption that they achieve annual average concentrations of dust of 8mg/Nm^3 and that efficiency gains of bag filters of 99% replaced existing ESPs of 95% efficiency.

One blast furnace reported needing to replace a filter to comply with the BAT conclusion. The cost of this was estimated to be €65,000 based on information reported by an EAF operator. The benefits were calculated using reported emission impacts for this furnace.

Two blast furnaces located in the same installation reported upgrading their bag filters and optimising flows treated by ESP. The cost for these two blast furnaces and for two blast furnaces that reported installing cast house dedusting systems was quantified using unit cost estimates from the I&S BREF. Specifically, the unit cost data were from the Austrian example on p.322 of the BREF, equivalent to CAPEX of €4.83m/t and OPEX excluding energy costs of €0.14m/t. These quoted cost data are from 1996, and per the footnote to Table 2-5, no uplift has been carried out to these costs. As the quoted OPEX of the Austrian reference plant in the BREF exclude energy costs, the estimated energy costs have been added using values from a Netherlands reference plant in the BREF and using non-residential electricity cost rates for 2015 for the relevant Member States from Eurostat (2018).

Table 2-29: BAT 61 - cost and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
6	6	31.9	2.3	2.3	PM_{10}	204	15.8

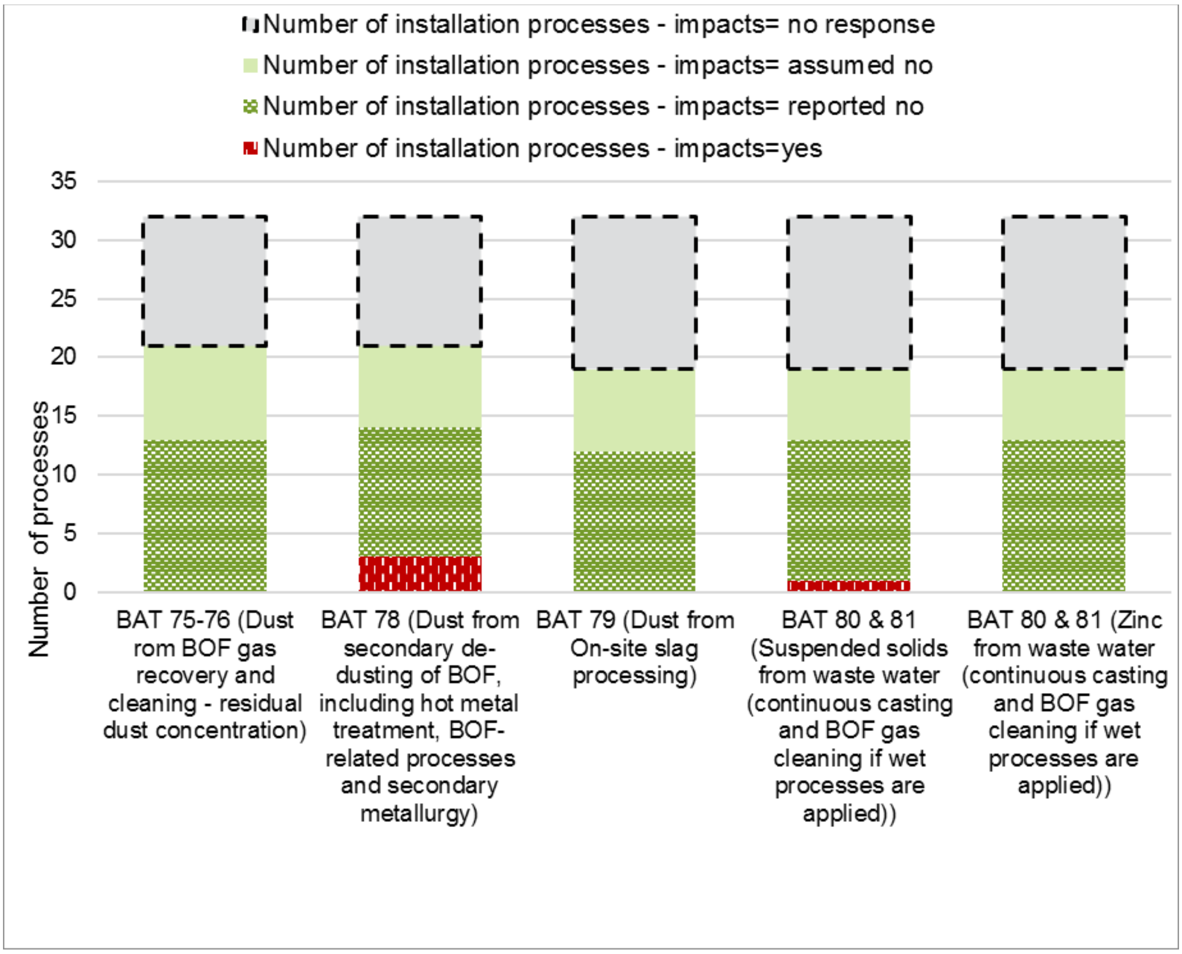
2.3.4.5 Basic oxygen furnaces

The BATs considered in the assessment were:

BAT 75-76 (Dust from BOF gas recovery and cleaning - residual dust concentration)
BAT 78 (Dust from secondary de-dusting of BOF, including hot metal treatment, BOF-related processes and secondary metallurgy)
BAT 79 (Dust from on-site slag processing)
BAT 80 & 81 (Suspended solids, zinc, iron, nickel, total chromium, total hydrocarbons from waste water (continuous casting and BOF gas cleaning if wet processes are applied))

The proportion of basic oxygen furnaces with information on BAT conclusion impacts is 59% of all BOFs in EU28, or 62% of the total EU28 capacity. Expressed in numbers, information was identified for 19 out of 32 basic oxygen furnaces. Only 3 of these have been identified as having been impacted by the BAT conclusions and 11 have reported no impacts. No impacts were assumed for an additional 5 installations based on responses from Member States regarding other steel processes but not mentioning BOF impacts. Two basic oxygen furnaces were identified as subject to a derogation under Article 15(4). No response was received from the remaining 13 BOF processes. A summary of the information identified is presented in Figure 2-12.

Figure 2-12: Information identified for BOF shop BATC impacts.



Note: the final column shown for BAT 80/81 represents not only zinc in waste water, but also iron, nickel, total chromium and total hydrocarbons.

Table 2-30: Summary of costs and benefits for BATs for BOF shops.

BAT	Number of processes impacted	Total annualised costs (€m/yr)	Benefits (€m/yr) of BATC compliance			Benefit-cost ratio
			Based on reported emissions	Based on estimated emissions	Total	
BAT 75/76	0	-	-	-	-	-
BAT 78	3	11.5	45.5	-	45.5	3.9
BAT 79	0	-	-	-	-	-
BAT 80/81 (suspended solids)	1	0.6	0*	0*	0*	-
BAT 80/81 (other pollutants)	0	-	-	-	-	-
Total	**3**	**12.2**	**45.5**	**-**	**45.5**	**3.7**

** No monetisation of suspended solids in waste water identified. The emissions quantity reduced is quantified.*

BAT 78 – Dust from secondary de-dusting of BOF

There were three installations with impacts for BAT 78: One reporting installation of a bag filter via the operator questionnaire (with reported costs and emissions data); one with a derogation in BAT 78 with a bag filter assumed to be installed by the end of the derogation (costs scaled from first installation), and one installation reporting only that secondary dust was reduced, which has been assumed to be via a bag filter. These three bag filters lead to assumed capital costs of €84.8 million, and annual PM_{10} emission savings of 832 tonnes valued at €45.5 million/year. In the absence of gas flow rate data for BOFs, emissions have been estimated by scaling reported emissions of other plants, which effectively assumes a fixed emission rate per tonne capacity.

A summary of the findings is presented in Table 2-31.

Table 2-31: BAT 78 - cost and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
2 (no derogation)	2	32.8	2.4	2.1	PM_{10}	847	19.4
1 (with derogation)	1	51.9	3.8	3.2	PM_{10}	1,341	26.0
Total	**3**	**84.8**	**6.2**	**5.3**	**PM_{10}**	**2,188**	**45.5**

BAT 80 & 81 – Waste water (continuous casting and BOF gas cleaning if wet processes are applied)

BAT 80 and 81 cover waste water pollutants of suspended solids, zinc, iron, nickel, total chromium and total hydrocarbons from continuous casting and BOF gas cleaning if wet processes are applied. Impacts were only identified for needing to meet the BAT-AEL for suspended solids in waste water for one operator. The cost impacts of BAT80/81 for this installation were the need to recirculate cooling water and water from vacuum generation. No monetisation has been possible for this impact.

Table 2-32: BAT 80/81 - cost and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
1	1	8.7	0.6	-	Suspended solids	1.1	-

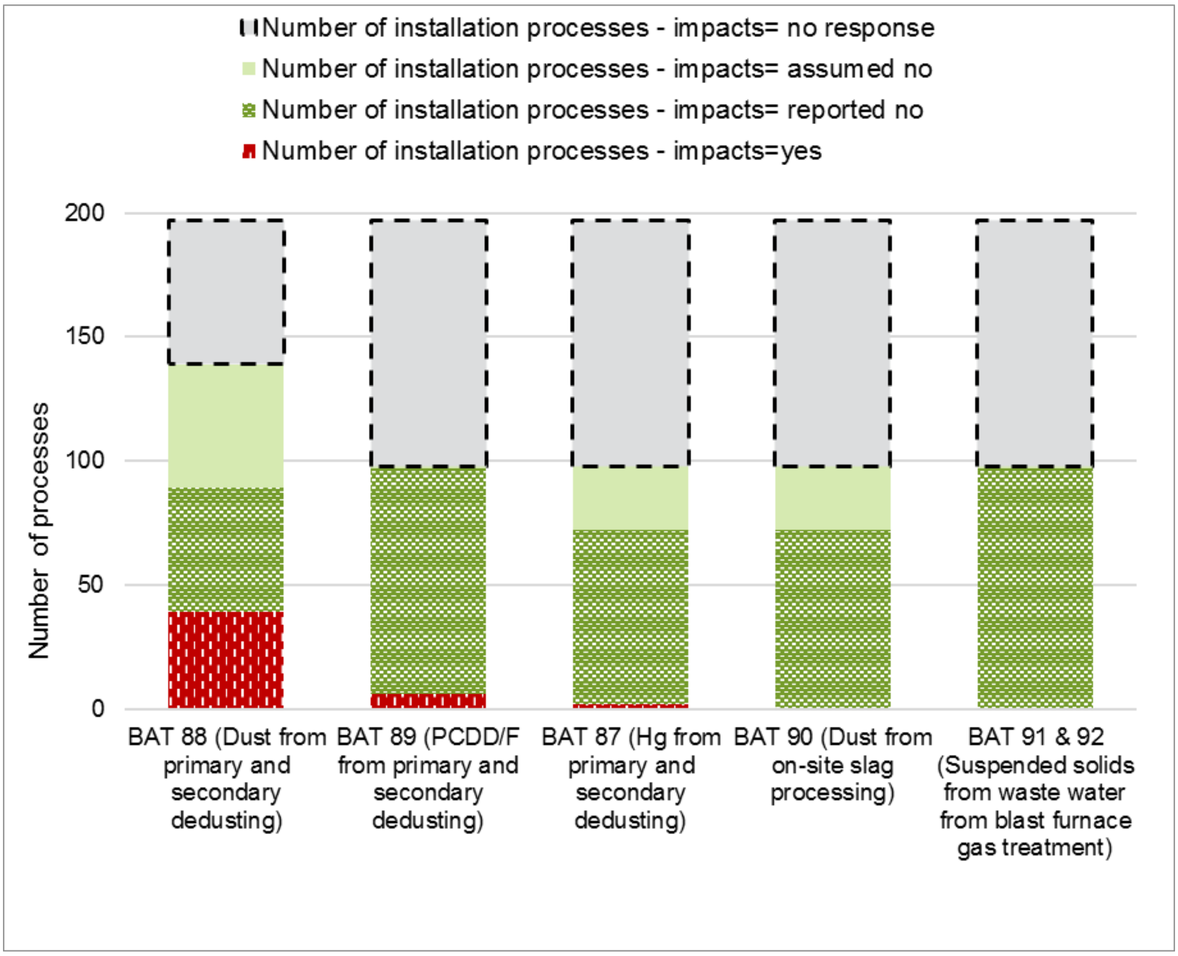
2.3.4.6 Electric Arc Furnaces (EAFs)

The BATs considered in the assessment were:

BAT 88 (Dust from primary and secondary dedusting)
BAT 89 (PCDD/F from primary and secondary dedusting)
BAT 87 (Hg from primary and secondary dedusting)
BAT 90 (Dust from on-site slag processing)
BAT 91/92 (Emissions of suspended solids, iron, zinc, nickel, total chromium and total hydrocarbons in the waste water from continuous casting machines)

Out of 197 identified EAFs in the database, there were 34 with reported impacts, 74 with reported no impacts, 11 installations where no impacts were assumed on the basis of information received from stakeholders, and a further 6 installations with impacts assumed on the basis of data within the VDEh database on currently installed abatement techniques. There are 72 EAFs without information on impacts. For this latter group of six EAFs there is an increased uncertainty in the impacts quoted. Costs have been taken from the BREF with regard to the technique "reduction of PCDD/F by means of adsorbent materials in combination with bag filters" for an EAF for the total off-gas flow, i.e. primary and secondary off-gases (NB this has a capital cost only and it is unclear whether the unit costs used represent the full costs that would be borne). A summary of the information identified is presented in Figure 2-13.

Figure 2-13: Information identified for EAF BATC impacts.



Note: the final column shown for BAT 80/81 represents not only zinc in waste water, but also iron, nickel, total chromium and total hydrocarbons.

Table 2-33: Summary of costs and benefits for BATs for EAFs.

BAT	Number of processes impacted	Total annualised costs (€m/yr)	Benefits (€m/yr) of BATC compliance			Cost benefit ratio
			Based on reported emissions	Based on estimated emissions	Total	
BAT 88	39	1.3	2.1	37.4	39.5	29
BAT 89	6	0.6	-	0.8	0.8	1.8
BAT 87	2	-	-	25.2	25.2	-
Total	**40**	**1.9**	**2.1**	**63.4**	**65.5**	**32.9**

** No monetisation of suspended solids in waste water identified. The emissions quantity reduced is quantified. BAT 87 costs not able to be estimated due to being around utilisation of different ferrous scrap feedstock; price paid for cleaner inputs is not known.*

Summary of impacts identified but not quantified for EAFs:

BAT	Impact	Aspect not quantified
BAT 88	24 EAFs with impacts identified of carrying out additional bag filter management	Capex, Opex, emission reductions- Lack of sufficient information to estimate impacts
BAT 87	2 EAFS with “utilisation of cleaner ferrous scrap”	Capex, Opex- No reported data on costs and not enough information to estimate

BAT 88 – Dust from primary and secondary dedusting

BAT 88 is the most significant BAT for EAFs in terms of costs and number of installations with impacts. Two EAFs report installation of better bag filters to replace their previous ones, with reported investment costs of €130,000. One EAF reported installation of a bag filter with investment costs of €3.7 million and annual benefits of €1.3 million. Six EAFs were assumed to install bag filters on the basis of not having BAT currently installed according to the VDEh database. This leads to total investment costs of €9.9 million and annual PM_{10} emission reduction benefits of €39.5 million per year. Comparison of annualised costs and benefits is shown in Table 2-34.

Co-benefits of PCCD/F emission reductions have also been estimated and included here, similarly to the method for sinter strands BAT20.

It was reported that some EAFs generally needed to implement improved bag filter management routines, however this impact was not monetised.

Table 2-34: BAT 88 impacts- Costs and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr unless specified)	Benefits (€m/yr)
39	18	9.0	0.7	0.6	Dust (primary)	732 (PM_{10})	39.5
(16)	*(16)*	*(Inc. above)*		*(Inc. above)*	PCDD/F (secondary benefit)	12.7 g T_{eq}	0.4

BAT 89 – PCDD/F from primary and secondary deducting

Information on PCDD/F impacts was available for four EAFs, provided by their operators. Capital costs totalled €2 million and provided emission reductions of 10.9 g, valued at €0.35 million. Three of the techniques involved carbon injection while one was described as a high temperature quenching system.

In addition, there are also co-benefits on PCDD/F from fitting of bag filters under BAT88, which is included for two installations under BAT88.

Table 2-35: BAT 89 - cost and emission benefits

No. of processes with reported impacts	No. of processes with reported impacts quantified	Capital costs (€m)	Capital costs (€m/yr)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
4	4	2.0	0.1	0.5	PCDD/F	10.9	0.35

BAT 87 – Hg from primary and secondary deducting

Two EAFs report the need to utilise cleaner ferrous scrap in order to reduce mercury emissions. No costs for this technique were available to be calculated.

Table 2-36: BAT 87 - cost and emission benefits

No. of EAFs with reported impacts	No. of EAFs with reported impacts quantified	Capital costs (€m)	Operating costs (€m/yr)	Pollutant	Emissions reductions (t/yr)	Benefits (€m/yr)
2	2 (emissions only)	-	-	Mercury	0.48	25.2

2.3.5 Results of sensitivity analysis

Sensitivity analysis was conducted on key assumptions that fed into technique cost estimates. The following tables show how sensitive the results are to changes in certain parameters:

- **The assumed lifespan of techniques**. The central results assumed techniques have 20 year lifetime, and thus their capital costs are annualised over this time period. The sensitivity of the result to a shorter assumption of 15 years is shown in Table 2-37.
- **The annual discount rate**. The central results assumed that capital cost repayments were discounted at a 4% (social) discount rate, in-line with European Commission guidelines. The sensitivity of the results to a higher discount rate of 10% reflecting possible private costs of borrowing is shown in Table 2-38.
- **The valuation of benefits**. The central results assumed an average of the VSL (high) and VOLY (low) approaches to valuing damage costs (see section 2.2.2.9). a sensitivity analysis of benefits was conducted to reflect uncertainties around damage cost benefit monetisation. The sensitivity results of 50% higher benefits than the central case (i.e. equivalent to the VSL approach) are shown in Table 2-40 and of 50% lower benefits than the central case (i.e. equivalent to the VOLY approach) are shown in Table 2-39.

The results of the sensitivity analyses summarised in Figure 2-14 indicate that while the results are sensitive to these parameters, even those with a greater effect do not change the overall conclusions, namely that the estimated impacts of the BATC have a positive benefit-cost ratio.

Figure 2-14: Summary of sensitivity analysis.

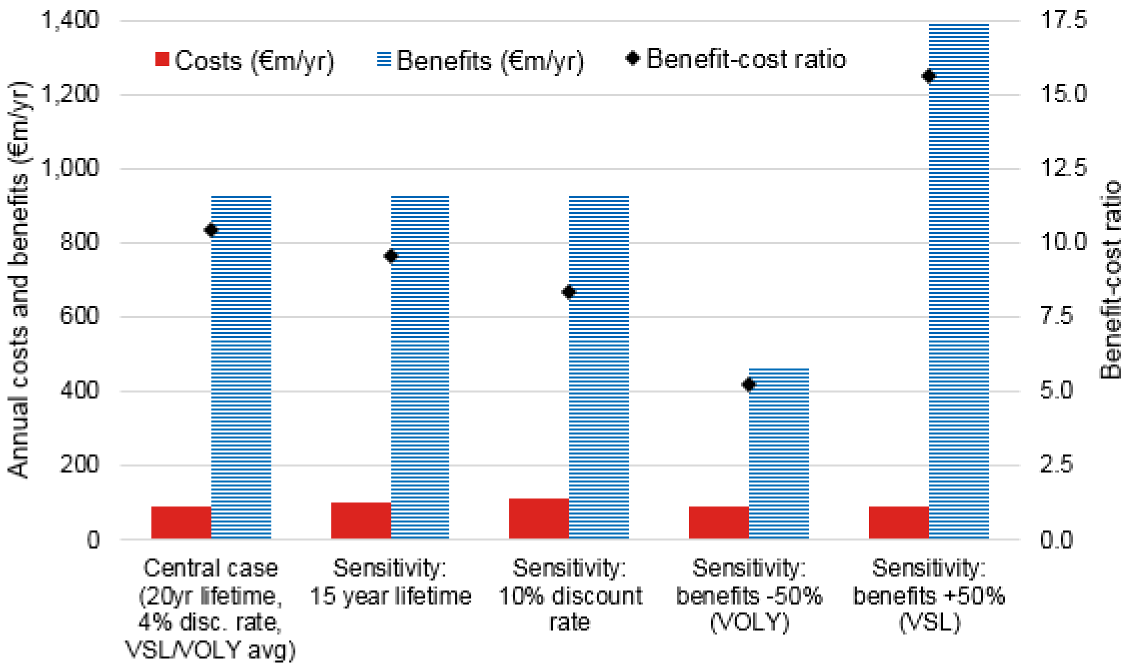


Table 2-37: Sensitivity analysis on technique lifetime - 15 year lifetime rather than 20 years

Process	Total annualised costs – central case (€m/yr)	Total annualised costs – sensitivity (€m/yr)	Increase in total annualised costs from central case (%)	Total benefits (€m/yr) of BATC compliance – central case	Benefit/cost ratio (central benefits /sensitivity costs)
Sinter strands	45.9	49.7	8%	644	13.0
Pelletisation plants	7.7	8.0	4%	6.8	0.8
Coke ovens	17.1	19.1	11%	155	8.1
Blast furnaces	4.7	5.3	11%	15.8	3.0
BOF	12.2	13.7	13%	45.5	3.3
EAFs	2.0	2.1	9%	65.5	31.1
TOTAL	**89.6**	**97.9**	**9%**	**932**	**9.5**

The results in Table 2-37 indicate that the increase in total annualised costs varies by process, (from 4 to 13%), which reflects how capital intensive on average the techniques are for each process. Assuming a reduction in technique lifespan by 5 years leads to an overall increase in total annualised costs of 9%, but this is not sufficient to change the study's conclusions on a highly positive overall benefit-cost ratio of the sampled two thirds of the sector.

Table 2-38: Sensitivity analysis on discount rate: 10% discount rate rather than 4% in central case

Process	Total annualised costs – central case (€m/yr)	Total annualised costs – sensitivity (€m/yr)	Increase in total annualised costs from central case (%)	Total benefits (€m/yr) of BATC compliance – central case	Benefit/cost ratio (central benefits /sensitivity costs)
Sinter strands	45.9	56.0	22%	644	11.5
Pelletisation plants	7.7	8.6	11%	6.8	0.8
Coke ovens	17.1	22.3	30%	155	6.9
Blast furnaces	4.7	6.2	30%	15.8	2.5
BOF	12.2	16.3	34%	45.5	2.8
EAFs	2.0	2.4	25%	65.5	27.2
TOTAL	**89.6**	**111.8**	**25%**	**932**	**8.3**

The results in Table 2-38 indicate that increasing the discount rate from 4 to 10% has a greater impact on costs than reducing the technique lifetime from 20 to 15 years: the costs increase by 25%. There is also substantial variation from process to process (11 to 34% increases). The increased costs in this sensitivity lowers the benefit-cost ratio, but does not change the overall strongly positive ratio for the studied two thirds of the sector as a whole.

Table 2-39: Sensitivity analysis on benefits: -50% benefits compared to central case

Process	Total annualised costs – central case (€m/yr)	Total benefits – central case (€m/yr)	Total benefits – low sensitivity (€m/yr)	Benefit/cost ratio (low sensitivity benefits /central costs)
Sinter strands	45.9	644	322	7.0
Pelletisation plants	7.7	6.8	3.4	0.4
Coke ovens	17.1	155	77	4.5
Blast furnaces	4.7	15.8	7.9	1.7
BOF	12.2	45.5	22.7	1.9
EAFs	2.0	65.5	32.8	17.0
TOTAL	**89.6**	**932**	**466**	**5.2**

The results in Table 2-39 show that reducing benefits by 50% (equivalent to using the VOLY approach) does not affect the overall strongly positive benefit-cost ratio for the two thirds of the sector covered in the analysis. However it does reduce the benefit-cost ratio specifically for the pelletisation process to drop below 1. As noted elsewhere the results for pelletisation processes are strongly dependent on the damage cost function for Sweden which is about one quarter of the EU average.

Table 2-40: Sensitivity analysis on benefits: +50% benefits

Process	Total annualised costs – central case (€m/yr)	Total benefits – central case (€m/yr)	Total benefits – low sensitivity (€m/yr)	Benefit/cost ratio (low sensitivity benefits /central costs)
Sinter strands	45.9	644	965	21.0
Pelletisation plants	7.7	6.8	10.1	1.3
Coke ovens	17.1	155	232	13.5
Blast furnaces	4.7	15.8	23.7	5.0
BOF	12.2	45.5	68.2	5.6
EAFs	2.0	65.5	98.3	51.0
TOTAL	**89.6**	**932**	**1,398**	**15.6**

Increasing benefits by 50% (equivalent to using the VSL approach) causes the already highly positive benefit-cost ratio to be even more positive. It also has brings the benefit-cost ratio for pelletisation process above 1.

2.3.6 Limitations of the cost benefit assessment and their implications

The assessment has required the use of a number of assumptions, and has also been rather limited by the response rate to the operator questionnaire. These assumptions add uncertainty to the estimated costs and benefits. The implications of the limitations on the results are summarised in Table 2-41.

Table 2-41: Limitations of the cost benefit assessment and their implications

Topic	Limitation	Implication on results
Sample size	There was a low response rate to the operator consultation. For many of the processes, information on the impacts of the I&S BATC has been identified for between 40-60% of the total population of the processes in the EU. The operator responses received provided a higher level of detail across a wider range of BATs than has been inferred from other material (Member State proforma responses, permit documentation, etc.).	The absolute values of costs and benefits may be underestimates of the total EU28 I&S sector impacts. It is possible that the installations without information on their impacts were not impacted. The benefit-cost ratios may however be sufficiently robust based on the sample size.
Emissions	There are some issues with the emissions estimate methods used which may lead to results being less accurate. Post-abatement emissions are used as a starting point for estimating emission reductions, with counterfactual pre-BAT emissions calculated from them.	In the few cases where achieved concentrations are reported and lower than the upper BAT-AEL, emissions reduction benefits are underestimated.
Emissions	The methodology relies on annual emissions and annual emission reductions. However, BAT-AELs and therefore also permit requirements for implementing BAT are usually based on short-term averages.	The CBA takes into account the long-term (annual average) approach. Benefits due to implementing BAT-AELs based on short-term (e.g. daily) averages are not considered. This might lead to the conclusion that implementation of BAT is not cost effective in some cases.
Benefits	Damage costs are averages for Member States and therefore might not reflect the local situation. As a consequence for some Member States, damage costs are rather low (of the order of 1/4 of the EU average).	If damage costs do not represent the local situation, this might lead to the conclusion that implementation of BAT is not cost effective.
Costs	It is unclear what costs would have been incurred under business as usual (i.e. a counterfactual scenario). For example, no account has been taken of the remaining economic lifetime of pre-existing techniques fitted at an installation process. The costs of technique renewal have not been deducted from the estimated costs of upgrades or replacement techniques. Such an assessment would ideally be undertaken but is not possible without more detailed and robust information on abatement techniques, particularly the dates when they were installed.	The estimated costs are likely overestimates of the true additional costs that would have been incurred beyond business as usual.

Topic	Limitation	Implication on results
Costs and benefits	Operators indicated that in some cases it was difficult to directly (and solely) attribute changes (investments) made at an installation to comply with the BATC. In many cases it was not specifically stated that techniques were implemented solely to comply with the BATC. In reality, operators' investments may have been driven by a number of factors, including tightened permit conditions due to the BATC, but also e.g. energy costs, market demand, other policies, local/regional situation.	The estimated costs and benefits are likely overestimates of those solely due to the BATC. This illustrates the difficulty of carrying out such cost-benefit assessments.
Costs	Capital costs have been annualised using an economic lifetime of techniques of 20 years and a discount rate (interest rate) of 4%. The private costs of borrowing capital may be higher for operators than a social interest rate of 4%.	For interest rates higher than 4%, the estimated costs will be underestimates. If technique lifetime is less than 20 years, the estimated costs will be underestimates. These are assessed in the sensitivity analysis.
Scaling unit cost data	Cost data from the BREF, and for a limited number of cases, reported costs from installation operators, have been used to estimate costs for another installation. This is carried out by scaling the cost according to capacity or waste gas flow rate (as appropriate). However, costs may not scale linearly with capacity or waste gas flow rate. Work by UBA (2002) has identified costs may scale according to $x^{0.7}$ where x is the scale factor.	Costs estimated (not reported) may not reflect actual costs; and could be under-estimates or over-estimates.
Emissions reductions	In a limited number of cases, reported achieved concentrations after installing a technique at one installation process are assumed to also be achieved at other installations.	This may not reflect actual achieved concentrations. The estimated benefits may be over- or underestimates.
Costs	Unless already included in the operating cost data, the results do not account for changes in energy costs that may occur if any of the techniques fitted would have led to increased energy consumption beyond the consumption of the existing installed techniques.	The estimated costs may be underestimates of the true costs.
Emissions reductions	In the cases where no reported data on the emissions impacts was available, emission impacts have been estimated where possible. To account for the uncertainty in the emissions estimates, two methods have been used (where possible) to estimate the emissions impacts.	Some benefits are presented as a range, reflecting the difference between two methods to estimating emissions impacts.

Topic	Limitation	Implication on results
Benefits	Damage costs are taken from EEA (2014) and Nedellec & Rabl (2016) and converted to 2015 prices. The damage costs do not monetise all health and environmental effects of the emissions of certain pollutants.	The estimated benefits are underestimates of the true benefits.
Benefits	The damage costs are a central value of a wide range (+/-50%) of uncertainty in benefit valuation. The range represents the different approaches to valuation of VSL and VOLY. A perfect valuation would use DCFs that account for the local level distinction between impacts from channelled emissions released through a high stack compared to diffuse emissions.	The benefits may be at least 50% higher or lower – sensitivity analysis has been conducted on this. Where benefit-cost ratios are above 2.0 it is more certain that benefits exceed costs.
Costs and benefits	The information about costs and benefits acquired was not always proportionate. For 3 installations, there was information only on costs but not benefits. In addition, for 33 installations there was only information on benefits but not on costs.	The estimated costs and benefits may be overestimated in these cases.

In addition to the table above, there are further uncertainties in:

- Assumed annual operating hours (8500 hours)
- Data from the VDEh database (e.g. capacity, flow rate). This has to a degree been addressed through validation checks undertaken on flow rates.
- Likely achieved emission concentrations.
- Pre-existing installed abatement techniques

2.4 Conclusions

2.4.1 Conclusions relating to the CBA of the I&S BAT Conclusions

A detailed process level cost benefit assessment has been carried out for the iron and steel sector to assess the impacts of the I&S BATC. Despite a low response rate to an operator questionnaire, a relatively detailed assessment has been possible for approximately two thirds of the sector. The assessment has overall estimated benefits to be significantly larger than costs.

Reflections on the process of carrying out the cost benefit assessment for the I&S BATC are:

- The high-level assessment that was carried out included three different methods to estimate the potential emission reductions attributable to the BATC. Using three rather than one method was chosen as it was unclear which method was the most accurate. **The key finding from the high-level assessment was that the three high-level methods encompass a very wide range in the predicted impacts on emissions for all air pollutants, leading to low confidence in predicted impacts.**
- Carrying out an initial high-level assessment leads to the generation of a hypothetical 'without BATC' scenario which is compared against a 'with BATC' scenario to generate estimated proportional reductions in emissions per pollutant. On the other hand, a detailed ex-post plant level assessment does not need to generate a 'without BATC' scenario and instead can estimate the benefits directly from estimated emissions reduced, and can estimate costs from

knowledge of techniques used based on stakeholder information. The high-level assessment carried out in this study is difficult to compare directly to the detailed assessment. However, a comparison showing changes in emissions between the high-level assessment and process-specific cost-benefit analysis is shown in Appendix 7. It shows that in the case of SO_2 and NOx, the CBA model estimates emissions reductions that are smaller than that of the high-level estimates. This may be because techniques were generally not reported by stakeholders for these pollutants and so benefits are underestimated in the CBA. In the case of dust, the CBA produces emission reduction estimates that are high in comparison to method 1 and method 2 of the high-level estimates (8 kt reduction compared with 1.3 and 4 kt respectively). Method 3 of the high-level estimates produced higher dust emission reductions (13.6 kt), but this is to be expected as it applies the concentration reductions from the BAT AELs to the entire sector rather than to specific installations as in the case of the CBA.

- Ex-post assessment relies on information being provided by operators and competent authorities, complemented by available collated datasets (public: EU ETS, E-PRTR) and private (VDEh). Limited information has been identified from other sources (permits, permit decision documents, operator publications).
- **More information is available on the impacts related to BATC impact on emissions releases to air than on impacts relating to emissions in waste water**.
- **Permits – and in particular permit decision documents – do include information that is useful for supporting a cost-benefit assessment of BATC compliance,** particularly relating to installation capacity, and techniques installed. However, few permits were identified or made available as few Member States make the permits easily accessible (inline also with EEB, 2017). Standardisation of permit structures would also make gathering information from permits easier. This should change with planned reporting under the IED for which the Member State authorities will have to report details of permits and where they are located.
- **Operators were generally reluctant to participate and provide data on impacts of the BATC.** Conducting a cost-benefit analysis of the impacts of the BATC on the iron and steel sector was made difficult by the (in general) lack of engagement of iron and steelworks operators. It is unclear whether this is a sector-specific issue or whether similar challenges would be encountered in other sectors. The iron and steel sector is made up of relatively few companies and installations however, which means that, compared to some other IED sectors, it should be easier to gather data for. Whilst it was suggested that a reason for low participation could be that operators may not be aware of the "BATC" as such (and thus have difficulty in identifying its impacts), and are rather more aware of their own permit conditions, this did not seem to be the case during the consultation carried out as part of this study.
- **The information provided can be difficult to validate as the information is unique to the installation or process in question.** There are only limited examples where there is evidence, from the Article 15(4) derogation process, that the regulator has considered and accepted/rejected costs quoted by operators as needing to be incurred to meet the updated BAT-based permit conditions, and in such cases, the costs can be assumed to be more certain. In other cases, there can be rather limited information available. For example, costs publicised by an operator as the value being invested in a certain technique or upgrade(s) due to BAT compliance must be accepted at face value, even if in fact that value may include installation upgrades that would have proceeded in the absence of the BAT conclusions. There may therefore be high uncertainty in the value that is directly attributable to the BAT conclusions.
- **Those installations granted a derogation according to Article 15(4) should generally have more information available on the impacts of the BATC.** This information is, at this juncture following the publication of the BATC, ex-ante information as it is a prediction of the emissions/cost impacts that will be incurred. Full cost data are not necessarily disclosed in such

permit decision documents (although what is presented is publicly consulted on); examples were identified that only made public the additional incremental cost between a scenario of BATC compliance by the four year deadline and a derogation scenario of delayed BATC compliance. However, due to the increased scrutiny on this information from it being a legal decision to derogate from BAT-based permit conditions, where information and/or data is available, there can be greater confidence in such data. Nevertheless, it should not be assumed that data for installations subject to the Article 15(4) derogation process is representative more widely of installations in the industry as by definition these are the installations for which the costs are disproportionately high compared to the benefits.

- **For many Member States, national-level competent authorities provided useful higher level data on the impacts of BATC such as the techniques installed to comply and at how many installations.** However, they lacked access to finer details on costs and emissions data. However, in some Member States, this more detailed information is only available at regional or local level, and this is time-consuming for national authorities to access and thus collate to contribute to such studies. It is also difficult for national authorities to suggest a sample approach to the identification of impacts – as identified as one option in Ricardo (2016) – as the national authorities are unable to indicate which installations could be considered representative. Possible solutions to this are the identification of the regional or local authority that has the highest density of the installation type in question, and contacting this authority directly. In the cases where national authorities are able to support studies assessing the impacts of the BATC, long timetables of up to four months should be provided to enable such data collection from subsidiary authorities.
- **It is difficult to ascertain retrospectively what changes would have been made at an installation under a hypothetical "counterfactual" scenario without the BATC**. For example, what investments would have occurred anyway to upgrade or replace existing techniques at the installation? This is because business do not normally consider hypothetical scenarios that reconsider decisions taken several years prior on when the techniques were upgraded or replaced. Based on the data available, it has not been possible to subtract from the information and data gathered on new investments made at installations the economic value in the remaining life of existing techniques or the costs of replacement of existing techniques. The costs that are quoted or estimated on techniques for compliance with the BATC therefore are expected to be overestimates. It is also difficult for operators to indicate what changes would have occurred had the lower BAT-AEL rather than upper BAT-AEL been implemented.
- **It can be difficult for an operator to retrospectively attribute investment costs to BAT compliance**. Operators make investment decisions for a variety of reasons. Aside from compliance with updated permit conditions following publication of BATC, an operator's decisions also take into account drivers such as energy prices, other policy requirements including environmental policies (such as EU ETS, or NECD), market competitiveness, corporate social responsibility (CSR). It is sometimes difficult for operators to specifically name the BATC as the key driver for their investment, and instead some operators have cited multiple drivers for their investment. Therefore it can be difficult for some operators to directly attribute costs, that they have incurred on investing in certain techniques, to needing to comply with the BATC. This will fundamentally affect the uncertainty of the availability of information available for ex-post assessment.
- **Secondary benefits** – i.e. reductions in emissions of additional pollutants to the target pollutant due to the installed technique – are important to take into account, as, depending on the pollutant, these can have a large impact on the overall cost-benefit assessment.
- **Process level data** - EU-level datasets such as E-PRTR provide information at installation level and not at process level. For estimating costs and benefits at the granularity of the BAT

conclusions, it was necessary to purchase access to a third-party database (VDEh) for critical data needed for estimating costs (e.g. capacity) and estimating benefits (e.g. flow rates). It was also useful in enabling certain assumptions such as existing installed abatement techniques. Such databases may not always be available for other IED sectors.

2.4.2 Conclusions relating to CBA of BAT conclusions

- **Reluctance of trade association** - EU-level trade associations have regularly complained to the Commission about the lack of cost benefit assessment in relation to BAT conclusions. This is despite the BREF reviews considering the costs of BAT.
- **Low proportion of processes impacted** – The evidence identified in the course of this work has shown a relatively small share of processes impacted by the BAT conclusions. This can mean that a high level of environmental protection was already being achieved and/or that there might have been room for BAT-AELs to be set at a more stringent level.
- **High level of benefit related to cost** – This assessment has indicated that overall the application of BAT in this sector appears to have led to a high level of benefit compared to costs.
- **Difficulty** – The difficulty to carry out such an ex-post CBA illustrates the impracticality of carry out such an assessment as part of the Seville process. These challenges arise for multiple reasons including the individuality of installations, the inability of operators to distinguish BAT related costs and their unwillingness to provide information (which may be due to multiple reasons).
- **Adequacy of Sevilla process assessment** – The high benefit to cost ratio in this ex-post CBA appears to support the approach followed of identifying and generalising BAT in the Sevilla process.
- **Greater difficulty in other sectors** – To carry out a comparable CBA for sectors other than Iron and Steel is likely to be even more challenging. This is firstly because the iron and steel sector is limited to a relatively small number of large installations, and secondly because a robust database was available covering in detail the processes in use at each installation. These conditions are unlikely to be replicated in most other IED sectors.

3 Identifying the most vulnerable parts of cost benefit analysis methodologies

This section presents the findings from identifying and assessing vulnerabilities which can drive variation in cost benefit analyses in relation to BATC (prioritised and supported with practical examples).

3.1 Approach taken

To identify vulnerabilities, several studies on CBA have been reviewed. The studies were selected by DG ENV. The following studies have been reviewed:

- Defra (2015) Air quality economic analysis. Damage costs by location and source
- Defra (2015) Valuing impacts on air quality: Updates in valuing changes in emissions of Oxides of Nitrogen (NO_X) and concentrations of Nitrogen Dioxide (NO_2)
- Andersen, M.S. & Brandt, J. (no date): Miljøøkonomiske beregningspriser for emissioner, Notat (methodology document explaining assumptions behind the damage cost functions in Denmark)
- Holland, M. (2017) Benefit assessment methodologies for the LCP BREF implementation. EEB published
- US EPA (2016) Supplemental finding that it is appropriate and necessary to regulate hazardous air pollutants from coal- and oil-fired electric utility steam generating units
- CEPS (2014) Assessment of Cumulative Cost Impact for the Steel and the Aluminium Industry
- Three confidential documents provided to Ricardo by the Commission, presenting analysis of costs of compliance with BAT-AELs in industrial sectors.

The studies were reviewed using a common template presented in Appendix 8. The template was designed to extract details of the methodologies for each key part of the CBA, specifically:

- defining the scope, baseline and counterfactual scenarios,
- calculations of emission reductions, costs and benefits, and
- presentation of final results.

The information extracted via the template was then assessed horizontally to identify sources of vulnerabilities in each part of the CBA. The assessment focused on the following four types of vulnerabilities: variation, omission, bias and error. This provided mapping of vulnerabilities against each analytical step. Based on the findings of the document review and own expertise in conducting CBA, the list of vulnerabilities was then prioritised using the likelihood of a vulnerability occurring, and its impact on the final results. The study review was most useful for informing the likelihood of vulnerabilities occurring rather than the impact on results. Given the availability and selection of case studies, it was not possible to make a systematic comparison of the impact of different approaches to individual aspects. Hence the assessment of impact was complemented by the experience of the project team. Using information extracted from the studies, practical examples of high-priority vulnerabilities were provided.

The final output is a prioritised list of vulnerabilities which can drive variation in CBA supported with practical examples. Given that only a small number of documents was reviewed, this was not intended to be a comprehensive list but a guide to where the variations are most likely to occur.

3.2 Vulnerabilities of CBA methodologies and sources of variation between national and EU-level assessments

In this section, the vulnerabilities of CBA methodologies and variations identified are discussed for each key part of the CBA in turn. The key parts draw on the previous work reported in Ricardo (2016) and include, for example, development of the baseline, the counterfactual scenario, and the step to calculate emission reductions.

3.2.1 Overall approach and scope

Aspect of the methodology	Vulnerability: Low High
Time horizon	
Perspective	
Level of analysis	
Sector coverage	
Geographical coverage	
Operator coverage	
Annualisation	

The **perspective** of the analysis (e.g. whether the analysis is undertaken from a societal, private, or other view) is identified as a key source of vulnerability. As observed in the study reviews, this can critically influence the coverage of impacts (and also the selection of data). Intrinsically the author of the study will select the approach which reflects their view and objective for the analysis. For example, societal analysis (typically undertaken by public institutions) will include consideration of both costs and benefits, which are usually environmental and/or societal and do not accrue to private firms, as in US EPA (2016). Studies undertaken by private firms also capture costs (and do so in a similar way) but typically omit consideration of environmental benefits (e.g. Confidential studies 1-3 and CEPS (2014)). As observed, this tends to impact more on the consideration of environmental or societal impacts, rather than costs. However, this can also affect impacts which accrue privately – for example fuel consumption may be an important benefit or cost – and may also affect the calculation of costs. Indeed both cases were experienced in the analysis of the Iron and Steel BATC. Unless already included in the operating cost data, the results did not account for changes in energy costs that may occur if any of the techniques fitted would have led to increased energy consumption beyond the consumption of the existing installed techniques. Furthermore, no account has been taken of the remaining economic lifetime of pre-existing techniques fitted at an installation process and the costs of technique renewal were not been deducted from the estimated costs of upgrades or replacement techniques – this could have led to the costs being overstated. Omitting key impacts and benefits from the assessment inherently biases the analysis against the option under consideration and could have a significant impact on the results.

As noted, perspective is intrinsically tied to the author of the analysis. Who is writing the report introduces wider **biases** than omission of impacts. Where CBA is conducted by those with a private commercial interest in the conclusions of the CBA (e.g. industry or affected operators), there is an inherent incentive to overestimate costs and understate benefits. This permeates both the overall approach and all stages of the analysis, in particular the selection of data, coverage, approach to emissions reductions and conclusions drawn. This was generally observed reflecting on the conclusions of study Confidential 2. When reviewing CBA studies, it is therefore important to consider the reasons for the study or its drivers, the incentives facing the author and how the results of the study could impact on the policy and its implementation. For example, US EPA (2016) was in part driven by challenge by the US Supreme Court that the US EPA had only to consider benefits, not costs. This may have placed pressure on them to over-state the costs, leading them to adopt the 'maximum cost' approach (see discussion on Time Horizon below).

Coverage of operators (i.e. which sites are included in the assessment, e.g. all or a sample) is also a potential key vulnerability. Coverage is highly likely to vary. For example, Confidential 1 looked at all operators in a sector, Confidential 2 considered a sample and Confidential 3 looked at hypothetical plants. By selecting a sample there is a strong risk that bias is introduced into the study. This may either be a conscious choice by the author or an unconscious one, for example where data limitations restrict the ability to define a robust sample (e.g. CEPS (2014) used a sample given a less than complete stakeholder response to a data request). Covering all operators is often not feasible and the choice of approach needs to balance available time, resources, and the size and variance of the sector. Where a sample is chosen, this should be representative of the sector and capture the range of operators although this is not guaranteed. Authors should present a clear case underpinning the selection of any sample to help alleviate concerns of bias. Using hypothetical plant could also introduce bias as this will omit consideration of outliers and raises the question as to whether they are representative of the population (Confidential 3 offers no explanation as to why hypothetical plants were used as the basis of the analysis). The size of this risk impacting on the results will depend on heterogeneity of the sector under consideration, but the impact of omitting / mis-representing part of the population could be significant. Targeting a full sample of sites could also introduce bias and should be sense-checked: for example, Confidential 2 seemed to estimate compliance costs for a plant that was deemed likely to have closed by the time BAT-AELs will come into force, artificially inflating investment costs. The assessment of the impacts of the Iron & Steel BATC had data for around two-thirds of installations and hence a relatively comprehensive sample. That said, the coverage of operators varied at a process level, as presented in Figure 2-7. Six out of seven pelletisation plants were captured in the sample – the highest coverage across the processes – and then only one of six plants were affected by the BATC. Hence the confidence in the representativeness of the sample is high. This is compared to coke ovens where only 26 of 53 coke oven plants were included in the study, of which 13 were affected by the BATC, suggesting there is a high probability that there are plants not in the sample that would have been affected by the BATC.

A less acute but still moderate vulnerability is **Time Horizon** (i.e. the length of the appraisal period). Although one would expect this to be fairly set in common CBA practice (i.e. horizon should be defined to capture all significant impacts) this is observed to vary somewhat in the study reviews. Again, there is an intrinsic link to available data – for example, the time horizon chosen by Holland (2017) reflected availability of the EMEP transfer matrix, a key input. Andersen and Brandt (no date) calculated external costs for 2013, using emissions data from 2008 and meteorological data from 2000. Where the horizon varies, streams of costs or benefits could be omitted from the analysis- hence this could significantly influence the impacts captured in the analysis, and the results. By design, the US EPA (2016) looked only at first year of impacts as this is when impacts were deemed greatest. In this case the US EPA did not use monetised cost-benefit analysis as a preferred method to compare the advantages and disadvantages of regulating hazardous air pollutant emissions from electric generating units. Instead the focus was on assessing compliance costs and this approach was designed to highlight the maximum level of costs (but if the study extended to compare costs to benefits this would not paint a fair reflection of the impacts observed over the full lifetime of the technology). The same analysis undertaken for the first year of impacts versus, for example, a 20-year time horizon would yield a different cost – benefit ratio. There is an intrinsic link between this vulnerability and the 'Lifetime' vulnerability explored under costs below – both will affect the timeframe over which impacts are captured. In addition, there is also a link to whether the study adopts a lifetime or annualised approach to assessing costs (as discussed below in this section). The discussion above focuses on where a lifetime approach is adopted. Under an annualised approach, as adopted in the analysis of the impacts of BATC on the Iron & Steel sector, the time horizon is still an important consideration and can still bias the comparison of costs and benefits. If the time horizon is too short, under a lifetime approach this would reduce the stream of future environmental benefits to compare to the costs. Under an annualised approach, reducing the time horizon increases the estimate of average annual costs relative to the average annual benefits. Hence in each case a shorter timescale can upwardly bias the costs relative to the benefits.

Level of analysis also presents a moderate vulnerability. Whether analysts select a 'top-down' or 'bottom-up' methodology[25] will depend on a number of factors, including resource available, data limitations (e.g. which required the US EPA (2016) to adopt a sector 'top-down' approach) or confidentiality (meaning data is unavailable at plant level). A high level estimate of impacts as conducted in this study can also inform the approach. However, as observed in CEPS (2014), parameters can vary widely at plant level hence any more aggregated approach inherently reduces the likely accuracy, relevance and pragmatism of the conclusions. The impact risk will depend on the relative coverage and the heterogeneity of the sector, which in turn will impact on the applicability of averages used in 'top-down' analysis.

Of lower concern are **sector and geographical coverage.** Both can vary as observed in the study reviews. In terms of sector, Confidential 1 looked at all similar technologies, Confidential 3 addressed a sub-sector and US EPA (2016) considered the whole power sector. In terms of geographical scope, some studies cover multiple operators in a single country (e.g. US EPA (2016)), some include multiple countries but a narrower subsector or number of operators in each country (e.g. Confidential 2) and some studies cover multiple operators and countries (e.g. Holland (2017) adopts a EU-wide scope). Again, coverage may be restricted by data availability (e.g. CEPS (2014) could only cover 1 out of 3 relevant sub-sectors). Although likely to vary, impact on results is deemed low as studies should focus on relevant sectors / geographies of interest, otherwise conclusions cannot be drawn. However, with such narrow approaches the results cannot be generalised.

Whether analysts adopt an **Annualised**[26] (e.g. US EPA (2016)) or lifetime approach (e.g. Confidential 2 which assesses all impacts over the lifetime of the technology) is also considered a lower vulnerability. This can often vary, and in some cases studies may present both (e.g. Confidential 3). The lifetime approach is typically viewed as a more robust and complete assessment, but an annualised approach is often adopted where analysts are limited in terms of data available or time and resource for the assessment. But where applied, this should still be justified as an appropriate approach by the assessor. In theory the two approaches should present similar conclusions, however this critically depends on how upfront costs compare to ongoing costs and how upfront costs are captured in any lifetime analysis. There will be a greater difference between a lifetime and annualised assessment:

1. The higher the significance of upfront costs relative to ongoing impacts
2. Where finance costs associated with upfront costs are not captured

In the case of abatement of industrial emissions, ongoing operating costs are typically more critical to BATC techniques relative to upfront costs, so the risk of this vulnerability is ranked lower than others. It is also important to note that when adopting the annualised approach, this implicitly assumes that the costs and benefits of the technique are fairly constant over the lifetime of the asset, and that the comparison between costs and benefits is representative of each year over the lifetime (and hence of

[25] I.e., 'top-down' refers to those which start from a sector or country-wide aggregation, relative to 'bottom-up' approaches which use data and undertake analysis at the facility or process level first, before potentially aggregating to a broader category such as sectoral or country level.

[26] Impacts, typically one-off upfront costs, may be annualised using lifetime of a measure and discount rate to provide estimation of the total annual cost of measure.

the lifetime as a whole). Where this is not the case, the conclusions drawn from the lifetime approach may diverge from those produced by adopting an annualised approach.

One final over-arching vulnerability is the risk of **errors**. From the study reviews, one observation was that errors could creep into UK air quality appraisal given the lack of specification of the coarse fraction to which particulate damage costs apply (although a subsequent error was not directly observed in the studies assessed). Generally, it is very difficult to identify errors as underlying calculations are not published and reports rarely include information on Quality Assurance or Peer review process.

3.2.2 Baseline scenario

Aspect of the methodology	**Vulnerability: Low**				**High**
Data sources					
Base year					
Compliance rate					

A key source of vulnerability in defining the baseline (and indeed across all the steps of the analysis) is the **data source** used. Different data is available to different organisations, data can be gathered from different sources and estimates for a single parameter in single year can differ widely for many reasons. Hence there is a high chance that data sources used could vary between studies, and the impact of using different data sources could be significant. Furthermore, data availability may vary by sector – indeed this was a key conclusion of the appraisal of impacts of BATC on the Iron & Steel sector. This appraisal was advantaged in that a high level of data was available, and available at BAT level, upon which the assessment could be based. It concluded that to carry out a comparable CBA for sectors other than Iron and Steel is likely to be even more challenging. This is firstly because the iron and steel sector is limited to a relatively small number of large installations, and secondly because a robust database was available covering in detail the processes in use at each installation. These conditions are unlikely to be replicated in most other IED sectors.

Data sources used can vary in the specific parameter used in the estimation. For example, Confidential 1 used a private database owned by a private company, Confidential 2 gathered data from operators via a survey as did CEPS (2014), Confidential 3 simply applied emission limit values from legislation and the US EPA (2016) used public databases. In no case did the review identify alternative, relevant sources which had been omitted. In several cases the source of underlying data was unclear, for example the supply of information about abatement measures in Confidential 1 and 2. The difference between using actual operating data relative to applying existing emission limit values has been explored further through the appraisal of impacts of BATC on the Iron & Steel sector. For the upgrading electrostatic precipitators at certain sinter strands to comply with BAT26 on secondary dust emissions, it was assumed for some processes that under the baseline a dust concentration of 48.5 mg/Nm^3 was achieved before BATC implementation and 20 mg/Nm^3 after BATC implementation. These assumptions were informed by reported data for a sinter strand. For another sinter strand these assumptions, together with the estimated flow rate, led to a baseline emissions prior to BATC implementation of 171 tonnes of dust, and after BATC implementation of 71t dust (reduction of 100t). If alternative assumptions had been made of baseline operation at the IPPC BAT-AEL of 50mg/Nm^3, and post BATC implementation at the IED BAT-AEL of 30mg/Nm^3, then the annual emissions calculations would have been 176t dust in the baseline and 106t after BATC implementation (reduction of 70 t dust).

Data sources can also vary in their robustness, which is tied to any underlying peer review and quality assurance process. For example, Defra's (2015) findings are based on extensive peer review of underlying health impact data by COMEAP, Andersen and Brandt's (no date) external costs have been subject to review by an inter-ministerial working group, and the model on which the US EPA's (2016) findings are based was developed and peer reviewed by a third party. On the other hand compliance cost data used in Confidential 1 were sourced from a private database which appears not to be independent and peer reviewed, and Confidential 2 used data provided by operators which cannot be independently verified. In the context of the analysis performed in this study, this raises the question of the robustness of the VDEh database. The inclusion of installations that have closed down or the absence of some operational installations suggests that there might be some issues with the quality of the data provided (see Section 2.2.2.2).

The data used in CBA can also vary in terms of disaggregation: Confidential 1 considered site-specifics around the techniques for application and Confidential 2 did likewise around emission levels. There can also be variance in the way multiple data sources and years of data are (and even if they are) combined, and the transparency of approach: CEPS (2014) used weighted averages from plant operators to calculate an average (this approach appeared appropriate, but this could only be verified by the technical expertise of the reviewer), whereas there is no information on how Confidential 1 combined monthly data to compare to annual averages (it seems to suggest these could be equal, in which case there may be a potential error). When data is published can also vary: for example, CEPS (2014) uses the latest published data, but Holland (2017) was limited to using 2010 data for the key EMEP matrix input.

There is no defined minimum level of data required that would facilitate (or even ensure) a 'successful' CBA. Whether the data available leads to a successful CBA or not will depend on the problem at hand, its nature and context. Furthermore, whether a CBA is successful of not will also depend on a wider range of factors, e.g. is method appropriate, has sufficient time and resource been devoted, are there any errors, etc.

Focusing specifically on data, whether the data available has facilitated or impeded a successful CBA being carried out will be a judgement, reflecting on the nature of problem and the appropriateness of the data available. For example, with the data that was available, what was 'known', what elements of the analysis were 'unknown', how were these unknowns addressed and how suitable were any assumptions used to fill gaps, how were conclusions drawn and how did these take into account any limitations in the underlying data. There is also a key link to the sensitivity analysis here, whether this has been performed and again how the results are reflected in the conclusions.

In general, the greater the quantity of and the more up-to-date the data available for the appraisal, the more likely the CBA will produce a robust assessment. But there is no reason why an appraisal based on limited data is automatically unsuccessful CBA: e.g. the sector may be very heterogeneous or small meaning data for a handful of sites would suffice, the sector may not change significantly over time meaning data published several years ago is still relevant, and/or the 'unknowns' may be unimportant or less significant in assessment such as where cost data is limited but environmental benefits are anyway large. Plus, there is no reason why access to substantial quantities of data leads to a successful CBA, for example this might not be used in right way again linking to the importance of other factors in determining the success of a CBA.

One factor which could bias results is assumed **Compliance rate assumed in the baseline** (i.e. the proportion of sites which comply with existing permits)**.** Across the studies reviewed, there is consistency in approach. For example, Confidential 1, 3 and CEPS (2014) all assume ELVs are currently met by plant, in lieu of actual performance data. That said, actual performance data should be public and compliance not confidential, although this may take greater resource to collate. Furthermore, Confidential 1 did not make any account for derogations. As such, where sites are already operating below ELVs, CBA could overstate (or understate where derogations apply) emissions and costs to comply with stricter regulations. This limitation is also noted as a caveat to the analysis in Holland (2017).

The **Base Year** (the starting point from which emissions, costs and impact calculations are made) adopted for studies is also a source of variance. However, typically studies will use the latest year for which key data is available (which as noted above could vary).

3.2.3 Counterfactual scenario

Aspect of the methodology	**Vulnerability: Low**				**High**
Counterfactual					
Projection factors					
Uptake of abatement technology					
Capturing impact of other policies					

Across all studies reviewed, none developed a future emissions scenario to use as a **counterfactual** (i.e. the 'without measures' scenario, in this case what the future would look like without the IED or BATC in place) suggesting that this is a less likely source of vulnerability. The counterfactual used in the studies assumed no future change in emissions compared to a baseline. Confidential 1, 3 and US EPA all left baseline emissions unchanged and did not apply adjustments to emissions factors (i.e. they did not adjust the emissions to reflect changes in the sector such as already planned investment in new abatement techniques). This is perhaps because forecasting introduces additional uncertainty into the assessment, and is often difficult – i.e. it is difficult to gather data on planned investments and other factors such as future growth are difficult to predict. Indeed, Ricardo (2016) concluded that projections of future emissions should only be carried out if robust data and assumptions are available. However, this could be important where other policies or market forces impact on the operations of affected sites in the future. Where a counterfactual is developed, this could introduce variation but where parameters are projected using the same factors in counterfactual and BATC scenarios, this reduces the potential to impact on the relative balance of costs and benefits.

Where studies develop a counterfactual, there may be variation in the **Projection Factors** adopted (i.e. parameters which are used to forecast variables into the future in the analysis). Often there is no single source which can provide an agreed best projection factor. However, multiple sources used to project the same factor are likely to be similar – e.g. where activity is projected using a growth factor, whether economic or population growth, both represent low percentage increases year on year. Furthermore, the projection factors applied could differ: for example, either activity growth or underlying efficiency changes (reflecting underlying **uptake of abatement technology**) could be varied, or both.

A further important vulnerability is whether analysts **Capture the impact of other policies** or not. Often the IED does not operate in isolation and there will be overlapping effects with other policy. Hence it is important to clarify what changes are associated with the baseline/other policy and which can be attributed to the IED. The US EPA (2016) did account for acid rain policy and CEPS (2014) assessed the combined effects of 8 policy areas, but the effects of IED were not split out (it is often very difficult to split out the effects of different policies which may have been a deciding factor in this approach). Other studies did not account for potential effects of other policies. This observation is consistent with the findings from this study. In some cases, operators reported that it was difficult to attribute investments solely to the IED and the requirements of the BAT conclusions.

3.2.4 Emission reductions

Aspect of the methodology	Vulnerability: Low				High
Selection of BAT-AELs					
Coverage of air pollutants					
Coverage of water pollutants					
Selection of techniques					
Partial or total impact of emissions abatement					

When estimating emissions reductions an important vulnerability is the **Coverage of pollutants, both air and water**. Typically the selection of pollutants will depend on the BATC of key concern, however general impact appraisal best practice dictates that analysts should identify all significant impacts. The studies did vary widely in terms of pollutants covered, but this does not in itself signal an error, more likely that there were different objectives and areas of concern between the studies. In no cases did the review find that studies had explicitly omitted pollutant impacts, however the review of Confidential 2 identified not all pollutants had been assessed for all plants. Hence although this is not a likely risk, it could have significant impact on the results where this vulnerability occurs.

It is worth noting that it is common for appraisals to be skewed more towards consideration of impacts through air pollution relative to water pollution. Although this issue is more related to the availability of methods and data to appraise impacts associated with pollution, rather than whether these are included in scope or not. This was illustrated in the appraisal of impacts of the BATC for Iron & Steel which concluded that more information is available on the impacts related to BATC impact on emissions releases to air than on impacts relating to emissions in waste water. In this case the BATC did include requirements related to water pollution which did cause issues for operators. However, a lack of data, in particular from monitoring prevented quantitative analysis in this case. Hence rather than omitting consideration of pollutants, this was more a case of limitations in methodology preventing estimation.

A key potential source of variance is the omission of secondary impacts which are a by-product of targeting the primary pollutant. This is typical where GHG emissions are associated with changes in energy consumption – Confidential 1 explicitly notes that secondary effects have been considered.

The **Selection of techniques** could also significantly impact on the results (i.e. how abatement measures are chosen for application in the analysis). The method of selecting techniques can vary across studies: Confidential 1 selected techniques on the basis of minimising costs, whereas Confidential 3 adopted measures which were deemed a 'typical' solution. Indeed, the justification for selection of techniques in many cases was unclear: Confidential 1 excluded some measures without providing sufficient justification, and Confidential 2 provided no justification for the techniques it deemed as required. In practice it is difficult to predict what measures will be put in place at which installations as the specifics vary on a site basis, however this should in part be informed by considerations of both cost and technical feasibility (the selection of techniques should be one aspect checked in sensitivity analysis given how difficult it is to select the techniques). In fact, the review identified potential errors in the techniques selection: in Confidential 2, the reviewer suggested that additional techniques may not be required in some plants. Furthermore, Confidential 3 made assumptions around the techniques already installed in certain types of plants which were deemed potentially inappropriate by the reviewer (Note: judging what technique is appropriate requires technical expertise and knowledge of the sector. A non-technical person would struggle to know if the selection of the technique is correct or not). This vulnerability predominantly relates to ex-ante assessments. For example, the appraisal of impacts of the BATC on Iron & Steel operators was largely unaffected as this was an ex-post analysis and as a minimum the underlying data provided evidence of the technique installed. In a few cases this information was not available and assumptions were made based on the wider dataset. Hence the extent to which ex-post assessments are affected depends on the availability of information on techniques installed. This is judged a medium-high risk as there is potential for the approach to and selection of techniques to vary between studies, with fairly significant impacts on the results if this is the case.

The outputs of the analysis are also vulnerable to the **Selection of BAT-AELs** – in terms of which BATC are considered, what point in the range of ELVs is adopted and the metric used for comparison. For example, Confidential 1, 2 and 3 all select the upper end of the range and focus on yearly (not daily) averages (in this case the selection of yearly averages could be justified as yearly averages can be stricter, meaning compliance with daily does not guarantee compliance with yearly. Again this selection should be justified by the assessor and sense checked by the reviewer for bias) . The study review hence suggested there is at least some consistency, but where the approach differs the results could be substantially impacted: this could significantly affect the choice of required techniques and costs to meet BATC.

Another important vulnerability is whether the studies assess the **Partial or total impact of emissions abatement** (i.e. are plant assumed to meet BAT-AELs exactly or the full impact of techniques). Partial assessments, where BAT-AELs are assumed to be met exactly, will almost certainly understate emission reductions and environmental benefits. For example, the US EPA (2016) adopted this partial approach, simply assuming operators comply with emission limits standards (in this case MATS, or mercury and Air Toxic Standards). Techniques will typically take operators beyond the BAT-AEL on a technical level. Operators also aim to operate below emission limits in order to have a buffer as part of their risk management strategy (Holland, 2017). In the review, Confidential 1 undertook a full assessment whereas Confidential 3 assessed the impacts of meeting BAT-AELs precisely. Estimates can also vary depending on whether the calculation is generic or specific for each plant or sub-sector: for example, Confidential 1 and US EPA (2016) adopt a generic calculation of reductions, whereas Confidential 3 assesses each hypothetical plant individually (but admittedly applying the same approach).

The study review also observed where biases could creep into the analysis, again due to the author of the analysis and the incentives they face. In Confidential 3, there is an assumption that some types of plants will not require any investment. The appropriateness of this assumption was questioned by the reviewer who went onto note it might have been made to make the results for other plants look less favourable.

3.2.5 Costs

Aspect of the methodology	**Vulnerability: Low**				**High**
Discount rate					
Lifetime of the technique					
Currency and base year					
Cost components					
Unit cost of techniques					
Secondary impacts					

In the estimation of costs, the selection of the **Discount rate** (i.e. the rate at which future impacts are reduced to reflect society or a private firm's time preference) can often be critical. This is inherently tied to the Perspective and objectives of the analysis. When undertaking a societal analysis, typically a 'social rate of discount' is adopted. For example, Holland (2017) adopted a societal discount rate of 4%, in line with common EU practice. However, even this study then deviated from common practice by adjusting the figure for future benefits to account for increased incomes and hence higher Willingness to pay (WTP) in the future[27]. This effectively reduced the discount rate to 2.4%. The US EPA (2016) adopted a relatively high discount rate of 7% for a societal analysis, but perhaps this reflects that it will be private operators investing. For the private studies, Confidential 1 and 3 adopted discounts rate of 5% and 10% respectively. 'Private discount rates' tend to be higher than societal ones given private sector borrowing costs are higher. High discount rates lead to higher annualised capital costs (or lower subsequent discounted benefits – whether discounting is applied as an uplift to costs or a reduction of future benefits, it has the same effect in the comparison of costs and benefits). Environmental benefits are typically discounted using a social time preference rate given they accrue to society, but where a private or higher discount rate is applied this reduces the value of future benefit streams. Indeed the review found that this assumption had a high impact on the results of Confidential 3. This is further illustrated through the sensitivity analysis conducted on the appraisal of impacts of the BATC for the Iron & Steel sector. The central results assumed that capital cost repayments were discounted at a 4% (social) discount rate, in-line with European Commission guidelines. The sensitivity of the results to a higher discount rate of 10% reflecting possible private costs of borrowing was tested and the results shown in Table 2-38. Increasing the discount rate increased the overall annualised cost from €89.6m/yr under the central case to €111.8m/yr under the sensitivity, reducing the cost-benefit ratio from 10.4 to 8.3.

Unit cost of techniques can also be a very important vulnerability for cost-benefit studies, both in terms of how cost inputs are selected from underlying sources (e.g. BREFs give a wide range of costs, so depending on whether the upper/lower cost end or an average value were selected, it could give very different results) and whether modifications are made compared to the original source. In practice, not many studies provide the necessary detail on how cost information is selected – in the review this was the case for Confidential 1, 2 and 3. Holland (2017) interestingly made some account for changes in costs over time: The potential for cost reductions as control techniques move from novel to mature is discussed, with potentially large reductions in early years but little change once technologies are mature. This vulnerability can be illustrated using the analysis of impacts of BATC on the Iron & Steel sector. In this case cost data was available in ranges and the mid-point was typically selected for the analysis. For estimating the costs of BAT61 (compliance for cast houses at blast furnaces), cost data were taken from the BREF. For this particular case, there were two cost sources quoted in the BREF. One of the sources was for a Dutch example, and in this example the opex costs quoted spanned a very wide range (€0.5/t to €2.8/t per year) for which no explanation was provided (e.g. it was not clear if the upper cost value also included disposal costs for dust, but which are not typical for this process). Given that energy costs were expected to be the largest component of the total opex costs, it was decided that these Dutch figures (and even the midpoint of €1.65/t per year) were not reliable enough to use. However, the second cost source was an Austrian example but in this source the quoted opex (~€0.14/t per year) explicitly excluded energy costs. It was decided to use this Austrian source and to add the energy costs is separately (which were estimated from the Dutch example energy consumption rate of 0.007GJ/t together with electricity prices), resulting in a final assumed opex of €0.32/t per year. This final assumed value is approximately one fifth of the mid-point of the Dutch example.

The assumed **Lifetime of the technique** is an important component of the analysis. Some variation is observed in the review (Confidential 1 assumed 10 years, Confidential 3 assumed 15 years and CEPS

[27] With higher expected income in the future, this reduces the value of money in the future relative to today.

(2014) 20 years) but evidently this will closely be associated with the techniques under consideration. The variation is thus likely a reflection of the different aims across the studies than in lifetimes of similar measures. Indeed, Holland (2017) notes that erroneous assumptions on plant lifetime could generate significant bias: too short a period will make it more difficult to demonstrate a net benefit, too long a period will make it hard to demonstrate a net cost. However, the likelihood of this occurring could be deemed low given the lifetimes of established techniques included in BATC will be fairly well known. As noted above, there is a link here to the Time Horizon for the analysis and Annualisation discussed in section 3.2.1 above. Time Horizon has a similar impact – this will impact on the length of costs and benefits captured in the analysis. regarding the annualised / lifetime approach, the lifetime of technique will still affect the analysis and in a similar way, but where the parameter feeds in varies. Under a lifetime approach, this defines the period over which impacts are captured – a shorter time period likely curtails the stream of environmental benefits to which costs are compared. Under the annualised approach, the lifetime of the plant feeds into the annualisation calculation determining the time period over which costs are annualised – a shorter time period increase the annual average costs relative to the annual average benefits.

There is uncertainty as to whether the lifetime of the technique or the lifetime of the plant is more relevant: typically lifetime of technique is selected, but lifetime of the installation should be adopted where this is shorter than (and hence limits) the lifetime of the technique (unless it is reasonable to assume the plant's lifetime will be extended, and the technique will achieve its full lifetime). There is also uncertainty whether the lifetime of the technique or of the policy should be considered – in the workshop one delegate suggested that a more suitable time period over which to assess the impacts of BATC is the BREF review period – i.e. the length of time before which requirements on plant could change again. That said, it is worth noting that policy timeline is more uncertain than that of the technique (i.e. it is uncertain exactly how long until the next review is initiated, and how long before the next updated BREF could be published), although the subsequent period of 4 years following BREF publication until permit conditions must be updated is known. Furthermore, when determining BATC commercial viability is taken into account – hence any consideration under a future BREF of changing or increasing requirements on operators who were required to implement changes under a previous BREF in theory should take into account any costs of stranded assets, hence reducing the likelihood particular techniques would only be in place for one policy cycle.

Sensitivity of the results to varying the lifetime of techniques has also been illustrated through sensitivity analysis performed around the assessment of impacts of BATC on the Iron & Steel sector. The central results assumed techniques have 20 year lifetime, and thus their capital costs are annualised over this time period. The sensitivity of the result to a shorter assumption of 15 years is shown in Table 2-37. The results indicate that the increase in total annualised costs varies by process (from 4 to 13%) reflecting how capital intensive on average the techniques are for each process. Assuming a reduction in technique lifespan by 5 years leads to an overall increase in total annualised costs of 9%. Although this is not sufficient to change the study's conclusions, it does reduce the cost-benefit ratio from 10.4 to 9.5.Studies could vary in terms of the **Cost components** (and sub-components) considered in the analysis. There are a range of different costs associated with compliance with BATC, e.g. upfront costs such as planning, site preparation or technology installation, and ongoing costs such as labour, fuel and maintenance costs. Best practice again dictates that all key impact categories should be captured, and typically studies capture a mixture of upfront and ongoing costs. Confidential 1 captured investment costs, including demolition, changes to plant, and connections, alongside operating and maintenance costs such as electricity consumption, compressed air use, waste treatment and disposal. Confidential 2 also captured total capex and yearly opex, including civils and project costs, ongoing filter replacement, etc. CEPS (2014) went beyond these studies to also capture administrative, financing and indirect costs. In some cases, studies explicitly note that some costs were excluded: Confidential 1 omitted wastewater treatment costs and Confidential 3 omitted the cost of intermittent operation associated with lower fuel quality. Although studies are likely to capture all key costs (in particular given studies written by private operators have an incentive not to understate costs), the inclusion or omission of impacts could be driven by data availability and in some cases could significantly impact on the conclusions. There is inevitably an incentive for any regulated entity to seek to maximise if not exaggerate the costs which it will face.

Currency and base year (i.e. the reference year in which monetary impacts are expressed) is a moderate concern. Specification of the base year (and perhaps inflation of impacts between years) is a factor often not explicitly addressed by studies - e.g. there is no information on Confidential 1, 2 and 3 regarding base year, and a 2012 reference was inferred from CEPS (2014) in the review. This could vary but adjusting for inflation between years should have only a very small impact, in particular if studies use fairly up-to-date data. The importance of this point is exemplified in the assessment of the impacts of the BATC for the I&S sector. For this assessment cost data was drawn from the BREF published in 1997. Updating to today's prices would have involved applying a high adjustment factor (around 1.58), significantly increasing the costs in the appraisal. On reflection, it was deemed inappropriate to apply such a large adjustment, in particular given the analysis does not take into account learning effects which would have placed downward pressure on costs over the same period as experience grew with installing and operating the technique. In the analysis, no adjustment was applied and the cost data from the original source assumed to apply in today's prices (this is explained further in the footnote to Table 2-5).

A less important vulnerability is **Secondary impacts** (i.e. impacts other than the key direct impacts of emissions reductions and costs)**.** Only one study in the review considered secondary impacts: US EPA (2016) assessed impacts of proposals on the market to provide power generating capacity. Best practice dictates that where quantification is not possible, primary and secondary impacts should be at least acknowledged and/or assessed qualitatively. Where these are addressed, these could signal important wider effects, but often these are difficult to quantify and include in the core CBA. Furthermore, some effects will be positive such as additional demand on equipment suppliers (but again these types of impacts are difficult to assess quantitatively).

As noted above, vulnerability to **data source** infiltrates all stages of the analysis. The review has identified instances where this affects the estimation of costs in the studies. Confidential 1 and 3 use the same private database for costs, but it is unclear how this has been developed, how independent it is, and what the underlying sources of data are. Cost data is difficult to verify as cost estimates inherently consider site-specific factors which cause costs to vary, and may also reference commercially confidential information (e.g. costs of capital). However, there are options available to verify cost information, including: comparing cost data to information available publicly (e.g. collected through the BREF process or wider literature), information available through the network of competent authorities (e.g. the Environment Agency in England and Wales are collating an evidence base of techniques and costs submitted through the derogation process to aid verification), or through engaging directly technique suppliers. Assessors could be asked to provide detail on how they came to their cost estimates and could also be asked to provide proof of quotes where feasible (e.g. where limited sites are being considered). Confidential 2 perhaps takes a more robust approach and draws cost data from technique suppliers, market data, and examples of other installations. However, even this information cannot be verified and detailed information on sources is limited. Likewise CEPS (2014) draws on a survey of operators whose evidence can also not be systematically verified, although comparisons are made to external publicly available data.

3.2.6 Benefits

Aspect of the methodology	Vulnerability: Low				High
Coverage and assessment method					
Assessment of health benefits					
Assessment of non-health impacts					
Transboundary impacts					

Studies often vary in their **coverage and the method** taken to assess the benefits of emission reductions. For example, Andersen and Brandt (no date) only consider public health impacts (both in Denmark and rest of Europe), and ignores all other impacts associated with air pollution (e.g. ecosystems, agriculture, materials, etc). Defra (2015) includes health impacts, and impacts on buildings and materials, but omits transboundary impacts, and impacts on ecosystems and crops. By comparison, Holland (2017) includes health and transboundary impacts, but also overlooks ecosystem impacts.

Again there is an inherent link to Perspective. As noted above, the review assessed several studies undertaken by private operators or their representatives which did not consider emissions benefits at all (Confidential 1, 2 and 3 and CEPS (2014)), introducing unavoidable negative bias in the results.

Even within categories of impacts, the range of effects included can vary. In terms of health impacts, Defra capture mortality and hospital admissions, but omit a range of conditions which also form part of WHO's HRAPIE (2013) guidance (e.g. chronic bronchitis, RADs, mRADS, etc). Whereas Holland (2017) adopts all HRAPIE's functions that are additive. But even this study admits to omissions, specifically NO_2-specific effects and a range of significant health impacts quantified separately by Nedellec and Rabl (2016). Andersen and Brandt (no date) refer to following the EC-CAFE assessment framework as the study pre-dates HRAPIE, but it is not clear exactly what impacts have been carried across to this study of external costs.

That said, the impacts of variance in the coverage of impacts can be significant. This is illustrated by the appraisal of impacts of BATC on the Iron & Steel sector which broadly applied damage costs developed by the EEA (2014). However, in the case of valuing mercury, damage costs were adopted from Nedellec and Rabl (2016) given concerns around the limited coverage of impacts contained in the EEA damage cost for mercury (further discussion on the coverage of damage costs can be found in Section 2.2.2.9). For example, the Nedellec and Rabl damage cost of €53,200/kg is applied in the assessment of BATC87: this is assessed to have achieved a reduction in mercury emissions of 0.48t/year, with a value of €25.2m/yr. Applying the lower EEA damage cost of €910/kg would have significantly reduced the estimated benefit to around €0.44m/yr.

Perceived limitations in the underlying evidence base are often the root-cause of disagreements on the inclusion of impacts. COMEAP (which informs Defra's assessment) tends to reach more conservative positions than the WHO. Its analysis and opinions are argued in detail, providing a clear rationale for their conclusions, suggesting there is a stronger and more certain evidence base underpinning these impact pathways. This does not, however, mean that the COMEAP conclusions are necessarily more accurate than HRAPIE, since the latter's recommendations were developed through detailed review of the evidence on health impacts of air pollution, conducted by a large group of invited experts from eminent institutions across the world. The aim of this work was to provide stakeholders with evidence-based advice on health impacts of air pollution and specifically recommend concentration–response functions for key pollutants should be included in cost–benefit analysis supporting the revision of EU air quality policy. In doing so, they specifically considered uncertainty and split impacts into categories depending on their ability to enable robust quantification of effects. Where health benefits are captured beyond HRAPIE, it would be prudent for the reviewer to check if the impact pathways were considered in the HRAPIE project, what the conclusion was, and whether estimation is based on any more recent evidence.

Lack of available evidence particularly affects the quantitative assessment of impacts of emissions to water. US EPA (2016) were the only study to calculate impacts of water pollution, assessing Intelligence Quotient (IQ) loss of recreational fishers. The study notes this to be small subset of the benefits of reducing mercury emissions. A wider set of impacts that are difficult to quantify are instead addressed qualitatively, including impacts on brain development, exposure through commercial fisheries, and non-health impacts such as on birds. Indeed this can also be an important variation – where impacts are not addressed quantitatively, key impacts should be identified and assessed qualitatively as these may play an important part in the conclusions drawn.

As noted above, given the limitations in methods and data available to appraise the health impacts of emission of pollutants to water, it is common for the appraisal of cross-media effects to implicitly focus on consideration of impacts through air pollution relative to water pollution. This was illustrated in the appraisal of impacts of the BATC for Iron & Steel which concluded that more information is available on the impacts related to BATC impact on emissions releases to air than on impacts relating to emissions in waste water (see section 2.4.1). In this case the BATC did include requirements related to water pollution which did cause issues for operators. However, a lack of data, in particular from monitoring prevented quantitative analysis in this case. Again it should be upon the appraiser to note the relative deficiencies in underlying methodologies and present the impacts in a way which represents their relative significance, whether assessed quantitatively or qualitatively.

Even where the same impact is assessed (e.g. mortality in Defra (2015) and Holland (2017), the specifics of the **assessment of health benefits** could also drive variation in the results. And there are a number of sources of potential vulnerability. First, the use and quality of dispersion models could impact on the results (e.g. Defra (2015) adopt a long established, peer-reviewed UK-focused dispersion model, whereas Andersen and Brandt (no date) apply an integrated regional-scale atmospheric chemistry transport model developed based on a Danish predecessor). Second, impacts could be assessed at different levels of detail and disaggregation: for example, Defra (2015) split by emission source and location (e.g. road transport, waste, agriculture), Andersen and Brandt (no date) apply a 'tagging-method' to calculate effects associated with different economic sectors (e.g. industry, agriculture, transport, waste, etc), whereas Holland (2017) produces averages per country (as this is a wider reaching study covering the whole of the EU). The impact of disaggregation on the results is evident from the analysis of the impacts of BATC on the Iron & Steel sector. That analysis adopted damage costs averages for Member States and therefore might not reflect the local situation. These damage costs offer an advantage over applying an EU-wide damage costs: for some the costs are of the order of 1/4 of the EU average hence applying a Member State specific damage cost where the site location is known avoids potentially overstating the benefits. However, a further issue is where the location of the plant is not known – i.e. for those not covered in the sample. For example, the pelletisation plant sample covered 6 out of 7 EU installations. The installations captured were predominantly based in Sweden and Finland, sparsely populated countries with subsequent low damage costs. It is questionable whether these damage costs are applicable to the seventh plant not captured in the sample, which could be located in a more populated country – in this case applying the Swedish or Finnish damage cost may understate the damages.

Third, studies could differ in the Concentrations Response Function adopted. Again Defra (2015) take their lead from COMEAP in this respect whereas Holland (2017) follows HRAPIE study for effects of primary and secondary particles and ozone, and the NEEDS study for toxic metals and other trace pollutants. (Note, however, that the recommended dominant response function, PM and chronic mortality is the same, 6% / 10ug.m-3 $PM_{2.5}$).

Fourth, the approach to monetising health impacts could also vary: Defra (2015) adopts a Value of Life Year (VOLY) approach whereas Holland employs both the VOLY and Value of statistical life (VSL) approaches to monetising mortality effects (monetisation of morbidity can also vary) to derive a range, and Andersen and Brandt (no date) simultaneously apply different approaches to different health effects (VOLY applied to chronic mortality and VSL to acute mortality). Fifth, even where valuation approaches are the same, the exact unit values can vary – e.g. Defra (2015) adopts a VOLY of £30,000 whereas Andersen and Brandt (no date) apply a VOLY of €75,800. By comparison, Holland (2018) applies a VOLY of €57,700 (2005 prices). Finally, there may also be variation in the source for baseline rates of health incidence and population, and how up-to-date this is. For example, Defra (2015) deploy the latest UK annual population statistics from the UK's Office of National Statistics whereas Holland adopts 2011 UN world population figures, and Andersen and Brandt (no date) adopt population data from 2000. There could also be variation in the method to estimate chronic effects.

As with health effects, there can also be variation in the methodologies adopted to **assess non-health effects** and the **inclusion or not of transboundary effects** (i.e. impacts of emissions outside the country of source). As noted above, Defra (2015) captures some non-health effects, namely impacts on buildings and materials, but does not capture impacts on crops or ecosystems nor transboundary effects. Holland (2017) does capture transboundary effects, using EMEP transfer matrices to cover the full EMEP domain for each country. Likewise, Andersen and Brandt (no date) also capture transboundary effects using a regional dispersion model. There is also high uncertainty in the valuation of impacts on ecosystems, as noted by the EEA (2014).

Reflecting collectively on these issues, one can observe a systematic bias in the Defra (2015) study towards underestimation – it takes conservative view on the range of impacts to include, lower unit values, omits ecosystem effects, etc. Even though Holland (2017) captures a wider range of impacts, they too still note a potential for underestimation given the omission of HRAPIE recommendations on NO_2 and lack of account of ecosystem impacts (the EEA report 'An updated assessment of the damage cost estimates to health and the environment caused by pollutants emitted to air from Europe's largest industrial facilities' published in 2014 (EEA, 2014) adopted the HRAPIE recommendations to assess health impacts, so perhaps can be considered more complete in this respect, but even this source it is said that "valuation of ecological impacts is currently considered too uncertain"). Although the mortality valuation takes alternative positions (VOLY, VSL) that give results that are both higher than the UK estimate, but are still lower than other estimates (e.g. made by the OECD (2012)). Andersen and Brandt (no date) too consider their estimates to be a conservative representation of the total health and all impacts of air pollution given omission of some emissions and impact pathways, and selection of conservative parameters for valuation.

3.2.7 Final results

Aspect of the methodology	**Vulnerability: Low**				**High**
Comparison of costs and benefits					
Output metric					
Uncertainty and sensitivity					

A key vulnerability between studies is the acknowledgement of **uncertainty and sensitivity** and (if and) how this is explored. Uncertainty is inherent in all cost-benefit analysis, driven by the data inputs and methodologies available, in particular in the estimation of environmental effects. Hence it is critical that analysts explore uncertainty and test the sensitivity of the results and the strength of conclusions drawn to underlying risks. However, many of the studies do not even acknowledge there may be uncertainty around their quantitative analysis (e.g. Confidential 2). In some cases, uncertainty is addressed in a limited way by simply discussing elements that could not be captured in the quantitative analysis (e.g. Confidential 3, and CEPS (2014) notes that impacts of the IED on electricity prices have not been included). Some studies go a step further and explore qualitatively and in detail potential uncertainties – for example Holland discusses key assumptions, the underestimation of benefits and presents a reference rule-of-thumb benefit-cost ratio for above which analysts can conclude that benefits outweigh costs, including those not captured in the quantitative analysis. Others undertake quantitative sensitivity testing around parameters, for example Defra (2015), Confidential 1 and US EPA provide ranges around central results by flexing key assumptions, such as benefit assessment and discount rate in the case of the latter. However, in some cases it is not clear why certain parameters are identified as key and flexed whereas others are not – e.g. CEPS (2014) presents an interval for opex but no other parameter. In others, it is not clear how uncertainty ranges were derived and what the ranges depend on (Confidential 1).

Studies can also vary in their **Comparison of costs and benefits** – e.g. by potentially omitting analysis from the overall results or drawing inaccurate conclusions. In the study review, where benefits were not assessed studies could not compare costs to benefits (e.g. Confidential 1, 2 and 3, and CEPS (2014)). Likewise, Defra (2015) and Holland (2017) only consider benefits so also cannot compare. The US EPA (2016) was the only study to compare impacts directly, and even then it was not a typical CBA given the focus was on assessing compliance costs (e.g. only a small fraction of benefits were monetised, and many more simply listed). Studies can also present their results differently by flexing the **Output metric**. Across the studies in the review, outputs were presented as:

- Environmental damages per tonne of pollutant (e.g. Defra (2015) and Holland (2017))
- Private costs per tonne of pollutant abated (e.g. Confidential 1)
- Cost per tonne of aluminium produced (CEPS (2014))
- Total investment / operating costs (Confidential 2)
- Net Present Value (US EPA (2016))

The metric presented will be tied to the Perspective and aims of the study. However, unless Net Present Value (NPV) is presented directly comparing costs to benefits, some form of reliable reference benchmark is needed to inform the judgement. The selection of a reference benchmark itself can introduce further vulnerability.

3.3 Conclusions on the most vulnerable parts of CBA methodologies

There are a wide range of vulnerabilities which could drive CBA studies to produce different costs and benefit outputs and conclusions, even where the same issue is being assessed. Table 3-1 presents a summary of the level of vulnerability caused by each aspect. In identifying the vulnerabilities and their likelihood and impact, this also helps to identify a list of checks which can be undertaken on the analysis. These can be prioritised according to the level of vulnerability assessed. These checks are presented in Table 3-2. In each case, the aspect extracted from the study should be sense-checked to see if it is robust, relevant and appropriate.

Table 3-1: Summary of vulnerability assessment

Vulnerability	Aspect
High	Perspective (Overarching) Operator coverage (Overarching) Data sources (Baseline) Coverage and assessment method (Benefits) Assessment of health benefits (Benefits)
High/Medium	Time horizon (Overarching) Coverage of air pollutants (Emission reductions) Coverage of water pollutants (Emission reductions) Selection of techniques (Emission reductions) Discount rate (Costs) Unit cost of techniques (Costs) Assessment of non-health impacts (Benefits) Transboundary impacts (Benefits) Uncertainty and sensitivity (Final results)
Medium	Level of analysis (Overarching) Compliance rate (Baseline) Counterfactual (Counterfactual) Capturing impact of other policies (Counterfactual) Selection of BAT-AELs (Emission reductions) Partial or total impact of emissions abatement (Emission reductions) Lifetime of the technique (Costs) Currency and base year (Costs) Cost components (Costs)
Low/Medium	Sector coverage (Overarching) Geographical coverage (Overarching) Annualisation (Overarching) Base year (Baseline) Projection factors (Counterfactual) Uptake of abatement technology (Counterfactual) Secondary impacts (Costs) Comparison of costs and benefits (Final results) Output metric (Final results)

Table 3-2: List of checks

Elements of the CBA methodology	Questions
OVERALL APPROACH AND SCOPE	
Time horizon	What time horizon has been assessed in the study? Has this been defined so as to capture all significant impacts? Has this been defined to present a fair comparison of costs and benefits?
	Has the rationale for the selection of the time horizon been explained? If yes, please summarise.
	Have any years been excluded? If so, why?
Perspective	Who is undertaking the analysis? What is the purpose / aims / objectives of the analysis? Could this create a risk of bias in the methodology, data or presentation of results? What stakeholders are the impacts assessed on (e.g. does the study consider cost and benefits to a private firm, society, government)?
	Have impacts on any stakeholder group been excluded? If so, why?
Level of analysis	Is the analysis conducted at the sector level (top-down) or at the plant level and aggregated (bottom-up)?
	Has the rationale for the level of analysis been explained? If yes, please summarise.
Sector coverage	What sectors, sub-sectors, products have been assessed? Are all affected sectors considered?
	Have any sectors / sub-sectors been omitted? If so, why?
Geographical coverage	What is the geographical scope of the assessment, e.g. EU / group of countries / Member State / region? Are all affected regions considered?
	Has any geography been excluded, if so why?
Operator coverage	Does the analysis cover all operators in each sector or sub-sector, a sample of operators or a single operator? Is the sample representative?
	Are any operators excluded, if so why?
Annualisation	Is the analysis presenting the annualised cost and benefits, or cost and benefits over lifetime? What is the balance between one-off, upfront impacts relative to ongoing effects?
Comments on potential bias	Is there anything in the overall approach and scope of the study that could suggest potential bias? Please describe.
Comments on potential errors	Is there anything that could suggest potential errors in this part of the CBA? Does the study describe the review process, quality assurance measures, or was the study peer reviewed? Please describe.
BASELINE	
Data sources (baseline)	What types of data sources were used? How was the data collected (e.g. literature, survey)?
	Have any sources of data been omitted? If so why?
	Are the data sources representative and obtained from independent providers? Can they be verified? Have they been peer reviewed?
	If multiple data sources were available, how was this dealt with? E.g. were averages, weighted averages calculated or a single, best source was selected
	If average values were used, have they been calculated using data from a single or multiple years?
	Have site-specific data been used? If so, for what parameters?
	When were the data sources used published? How recent was the data used?
Base year	What is the base year for the assessment? Does this represent the latest year for which data is available? Has the selection been justified?
	Is the study likely to under / overestimate emissions in the base year? E.g. through choices on the data and averaging used.
Assumption on compliance rate in the baseline	What is the assumed compliance rate with the permit conditions? E.g. are all installations assumed to meet current permit conditions or is the actual performance data used instead? Does this reflect derogation?
Comments on potential bias	Is there anything that could suggest potential bias in this part of the CBA? Please describe.
Comments on potential errors	Is there anything that could suggest potential errors in this part of the CBA? Does the study describe the review process, quality assurance measures, or was the study peer reviewed? Please describe.

Elements of the CBA methodology	Questions
COUNTERFACTUAL	
Counterfactual	Has the counterfactual scenario assuming no BAT-AEL impact been developed?
	If this step was omitted from the assessment, why?
Projection factors	What projection factors have been Applied? What data source has been used?
	How were the projection factors selected (e.g. assuming no / low or high future growth)?
Uptake of abatement technology	What is the assumed uptake of the abatement technology?
Capturing impact of other policies	Have impacts of other policies have been captured?
	How have impacts of other policies been isolated from the impact of IED?
	If any policies that could influence the results been omitted, why?
Data sources (counterfactual)	What types of data sources were used? How was the data collected (e.g. literature, survey)?
	Have any sources of data been omitted? If so why?
	Are the data sources representative and obtained from independent providers? Can they be verified? Have they been peer reviewed?
	If multiple data sources were available, how was this dealt with? E.g. were averages, weighted averages calculated or a single, best source was selected
	If average values were used, have they been calculated using data from a single or multiple years?
	Have site-specific data been used? If so, for what parameters?
	When were the data sources used published? How recent was the data used?
Comments on potential bias	Is there anything that could suggest potential bias in this part of the CBA? Please describe.
Comments on potential errors	Is there anything that could suggest potential errors in this part of the CBA? Does the study describe the review process, quality assurance measures, or was the study peer reviewed? Please describe.
EMISSION REDUCTIONS	
Selection of BAT-AELs	Is the method assuming compliance with the higher or lower BAT-AEL? If available in the study, summarise why.
Coverage of air pollutants	What air pollutants are covered? Does the analysis capture secondary impacts where relevant?
	Are any pollutants explicitly omitted? If so why?
Coverage of water pollutants	What pollutants are covered? Does the analysis capture secondary impacts where relevant?
	Are any pollutants explicitly omitted? Why?
Selection of techniques	How were the abatement techniques selected?
	What techniques were selected and is the selection justified?
Partial or total impact of emissions abatement	Are operators assumed to meet BAT exactly? Or is the study determining the true, full impact of application of techniques?
	Has the assessment of emission reductions been done for each plant / sub-sector specifically or was a generic emission reduction factor used across all plants / sub-sectors?
Data sources (counterfactual)	What types of data sources were used? How was the data collected (e.g. literature, survey)?
	Have any sources of data been omitted? If so why?
	Are the data sources representative and obtained from independent providers? Can they be verified? Have they been peer reviewed?
	If multiple data sources were available, how was this dealt with? E.g. were averages, weighted averages calculated or a single, best source was selected
	If average values were used, have they been calculated using data from a single or multiple years?
	Have site-specific data been used? If so, for what parameters?
	When were the data sources used published? How recent was the data used?
Comments on potential bias	Is there anything that could suggest potential bias in this part of the CBA? Please describe.
Comments on potential errors	Is there anything that could suggest potential errors in this part of the CBA? Does the study describe the review process, quality assurance measures, or was the study peer reviewed? Please describe.

Elements of the CBA methodology	Questions
ASSESSMENT OF COSTS	
Discount rate	What discount rate has been assumed?
Lifetime of the technique	What lifetime of the technique has been assumed?
Currency and base year	What currency and base year for costs are used?
Cost components	What types of costs are included (e.g. investment costs, maintenance costs and the most significant operating costs)? Are financing costs included? What cost components have been omitted?
	What are the sub-components of the key cost categories (e.g. what is assumed as part of the maintenance, operating costs etc.)?
Unit cost of techniques	How were the cost inputs selected? E.g. BREFs give a wide range of costs, so depending on whether the upper/lower cost end or an average value were selected, it could give very different results
	What modifications have been made to cost values compared to the original source?
Data sources (costs)	What types of data sources were used? How was the data collected (e.g. literature, survey)?
	Have any sources of data been omitted? If so why?
	Are the data sources representative and obtained from independent providers? Can they be verified? Have they been peer reviewed?
	If multiple data sources were available, how was this dealt with? E.g. were averages, weighted averages calculated or a single, best source was selected
	If average values were used, have they been calculated using data from a single or multiple years?
	Have site-specific data been used? If so, for what parameters?
	When were the data sources used published? How recent was the data used?
Secondary impacts	Have any secondary impacts of the costs been assessed, e.g. on business affordability, supply chain, competition were assessed? If yes, how?
	If secondary impacts were not assessed, does the study explain why?
Comments on potential bias	Is there anything that could suggest potential bias in this part of the CBA? Please describe.
Comments on potential errors	Is there anything that could suggest potential errors in this part of the CBA? Does the study describe the review process, quality assurance measures, or was the study peer reviewed? Please describe.
ASSESSMENT OF BENEFITS	
Coverage and assessment method	What benefits are assessed (e.g. GHG reductions, health benefits) and are they assessed quantitatively or qualitatively?
	What benefits are omitted from the study and why? Are effects assessed qualitatively that cannot be assessed quantitatively?
Assessment of health benefits	How are emissions impacts on concentrations and exposure modelled?
	Is the modelling disaggregated by the type of emission source?
	What health impacts are covered (e.g. mortality, asthma, lung cancer, hospital admissions, chronic bronchitis, work day lost, reduced activity days)?
	What health impacts are excluded? What aspects of the valuation has been omitted?
	What is the source for the Concentration Response Functions (CRFs) used (i.e. underlying epidemiological studies)?
	What valuation approach has been used for mortality effects (e.g. Value of Life Year Lost (VOLY) or Value of a Statistical Life (VSL))?
	What valuation approach has been used for morbidity effects (e.g. willingness to pay, resource cost)?
	What unit impact values have been applied (i.e. please extract specific values for the VOLY, VSL etc depending on what approach was taken in the study)
	What is the baseline rate of health incidents (e.g. number of deaths per year, number of hospital admissions that you get per year)?
	What is the baseline population?
	How are chronic effects taken into account (e.g. first year of impact or impact over life)?

Elements of the CBA methodology	Questions
	Is there anything that has not been covered in the list of vulnerabilities?
	Have low/high bound estimates been used to exaggerate/downplay the results?
Assessment of non-health impacts	How have the non-health benefits been assessed?
Transboundary impacts	What transboundary impacts are captured?
	How were the transboundary impacts captured?
Comments on potential bias	Is there anything that could suggest potential bias in this part of the CBA? Please describe.
Comments on potential errors	Is there anything that could suggest potential errors in this part of the CBA? Does the study describe the review process, quality assurance measures, or was the study peer reviewed? Please describe.
FINAL RESULTS	
Comparison of costs and benefits	How are the costs compared to benefits? Are the conclusions drawn reflective of the results presented? Do they take into account both quantitative and qualitative evidence, and any uncertainty around the analysis?
	Have any results been omitted / not presented or considered in the final assessment? If so, why?
Output metric	What output metric has been used (e.g. EUR/tonne of pollutant reduced; EUR/tonne of product manufactured)? Is a reference used to which to compare the results? Is this relevant?
Uncertainty and sensitivity	Has uncertainty in the analysis been acknowledged and explored?
	Are results presented as a range given the uncertainties? If no, how is a single figure selected for the results?
	Has sensitivity of results been tested against key assumptions / parameters? If so, which parameters and assumptions were selected for the analysis
	Are elements that could not be captured as part of the quantitative analysis clearly stated?
Comments on potential bias	Is there anything that could suggest potential bias in this part of the CBA? Please describe.
Comments on potential errors	Is there anything that could suggest potential errors in this part of the CBA? Does the study describe the review process, quality assurance measures, or was the study peer reviewed? Please describe.

4 Moving towards best practice in damage costs estimation at national level

4.1 Introduction and approach taken

This section describes best practices and actions required to produce air pollutant damage costs at national level. The key steps carried out for this task were:

- Define what actions and activities are required to develop damage costs at a national level
    - Consideration of the overarching principles that should guide the development of national damage costs (such as coverage of transboundary impact)
    - Step-by-step assessment of the impact pathway approach to understand whether and how each step could be changed to make the assessment nation-specific.
    - Consideration of the tools and models required to produce national damage costs.
- Review the methods and data used to derive the EEA damage costs

The output of the task is a series of recommendations on best practices and actions required to produce air pollutant damage costs at national level with information on the resources needed, time required and timeframe, expertise and costs.

4.2 Overarching principles

The quantification of the damage associated with pollutant emissions, and the benefits of pollutant control need to be seen against:

- The polluter pays principle[28], with overlap to the single market
- The preventive and precautionary principles
- Multi-media concerns
- Wider environmental and social goals
- The EU's Better Regulation agenda.

4.2.1 The polluter pays principle

The Polluter Pays Principle was first mentioned in 1972, and has been a feature of European action since the first Environmental Action Programme (1973 to 1976). It is defined under Principle 16 of the UN Declaration on Environment and Development as follows:

> *"National authorities should endeavour to promote the internalisation of environmental costs and the use of economic instruments, taking into account the approach that the polluter should, in principle, bear the cost of pollution, with due regard to the public interest and without distorting international trade and investment."*

Distortions of international trade can arise in two ways:

1. By excessive application of the principle, or more commonly,
2. By not internalising externalities, or by paying subsidies to polluters, for example to pay for preventive measures or to support older polluting industries.

The polluter pays principle does not set boundaries in terms of who or what may be considered a relevant receptor of damage. On this basis, impacts should be accounted for wherever they occur, not only in the area surrounding a plant, or the country that the plant is located in, but across the whole EU

[28] Training package on principles of EU Environmental Law. http://ec.europa.eu/environment/legal/law/principles.htm

and beyond. This is recognised and accepted in global legislation, for example addressing climate change or persistent pollutants such as mercury and various pesticides. However, it is not always reflected in attitudes towards regional and continental scale problems, such as air pollution, as will be shown below.

Similarly, the polluter pays principle does not specifically set boundaries in terms of some arbitrary definition of acceptable levels of damage. From an economic perspective, 'scarce resources' are optimally deployed when marginal costs and benefits of pollution are equal. Less abatement would lead to an excess of damage over cost, and vice versa.

The polluter pays principle is consistent with the single market, in setting uniform environmental and health goals.

4.2.2 The preventive and precautionary principles

Principal 21 of the Stockholm Declaration of 1972, and Principle 2 of the Rio Declaration on Environment and Development is as follows:

> *'States have … the responsibility to ensure that activities within their jurisdiction or control do not cause damage to the environment of other States or of areas beyond the limits of national jurisdiction"*

The principle of prevention is further included in numerous other Conventions and in EU law (including, for example, the Industrial Emissions and the Seveso Directives). Similar to the discussion above on the 'polluter pays principle', it asserts the need to account for damage at source, wherever it occurs, and not just within one's own national boundaries.

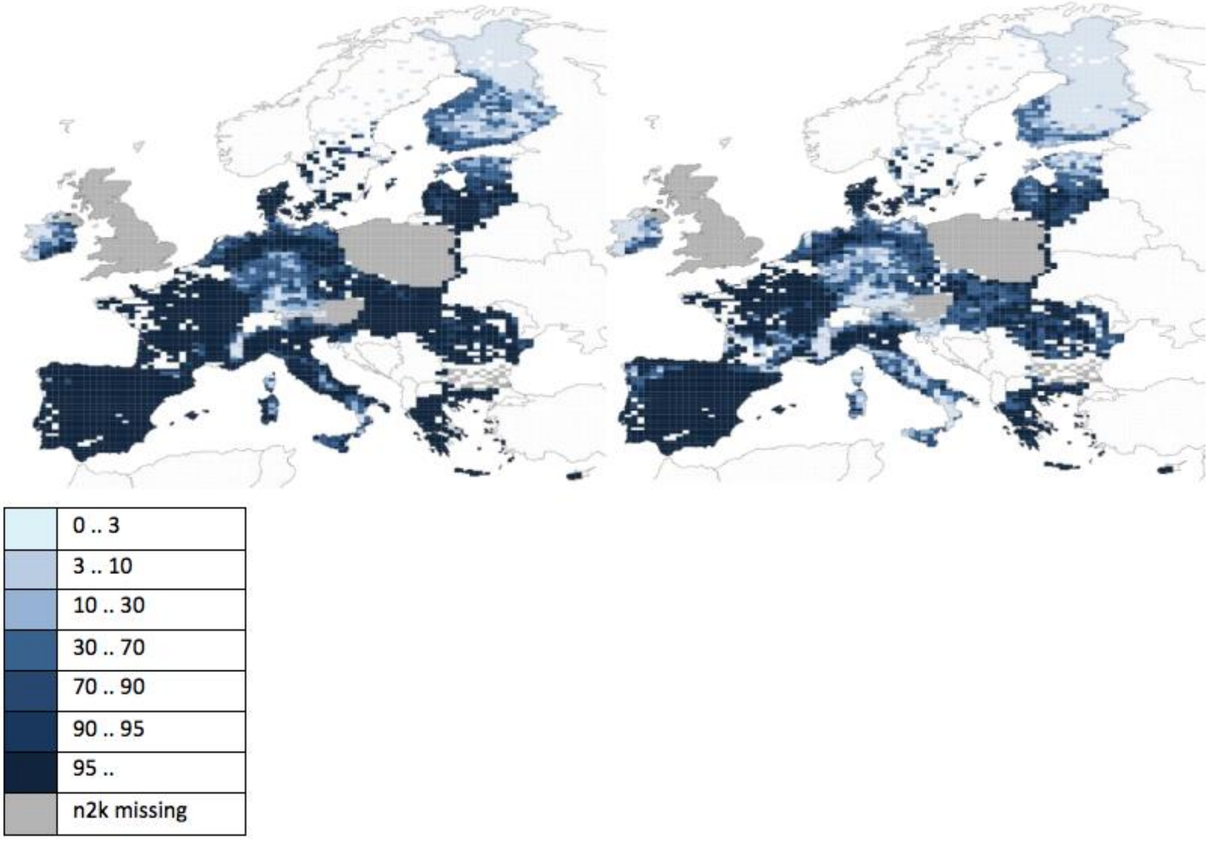
The precautionary principle is defined as Principle 15 of the Rio Declaration:

> *"In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation."*

The role of the precautionary principle is seldom debated with respect to air pollution though there is extensive discussion of the extent to which it should be applied in development of environmental regulation. One reason for this limited debate on the application of the precautionary principle for air pollution is that it has been possible to quantify health effects of the major pollutants (NH_3, NOx, PM, SO_2, ozone) with a reasonable level of confidence to inform the strengthening of policy. However, there are areas where it may be considered further, for example in relation to:

- Extensive exceedance of critical loads for nitrogen, as highlighted in the EU's Clean Air Outlook of 2018 (European Commission, 2018) and illustrated in Figure 4-15. There is forecast to remain extensive exceedance of critical loads for eutrophication, even with full implementation of the National Emission Ceilings Directive by 2030. This may pose a threat of irreversibility if exceedance is maintained for a long period, leading to significant ecological change.
- Toxic impacts of metals and other trace pollutants beyond those that are typically quantified. Nedellec and Rabl quantified much higher damage costs for lead, arsenic, mercury and cadmium than earlier estimates by bringing in new evidence that enabled them to significantly expand the range of health effects considered.

Figure 4-15: Percentage of ecosystem area with nitrogen deposition above critical loads for eutrophication (2005 left, vs full implementation of NECD 2030, right). Source: EU Clean Air Outlook 2018.



4.2.3 Multi-media concerns

Discussion in this chapter focuses on air pollution because its effects are now routinely quantified. However, concern also applies to discharges to water and land, recognising the nature of Integrated Pollution Prevention and Control on which the IED is based. Economic analysis of impacts via these media is very limited, especially in a European policy context. Damage tends to be very site specific, making generalisation difficult or impossible. Potential methods for quantifying effects of liquid discharges were discussed in an earlier report to the Commission (Ricardo, 2016). Further work is ongoing in this area under the BLUE2 study [29] which has the following objectives:

- To identify the economic benefits of EU water policy via the Water Framework Directive, and the cost of its non-implementation. Two elements are considered: the value of clean water as an input for the European economy, and more generally the value of a healthy freshwater environment.
- To build up a Europe-wide capacity for integrated assessment modelling of policies that affect the quality of the freshwater and marine environment.

BLUE2 is scheduled to be completed by the end of 2018.

4.2.4 Wider environmental and social goals

There are strong links between policy on air pollution and other EU policies. One of the most direct is with climate legislation, given that local and regional air pollutants share a major common source with

[29] http://ec.europa.eu/environment/blue2_en.htm

greenhouse gases, through the combustion of fossil fuels. This highlights a need to consider not only the direct, intended, consequences of legislation, but also ancillary impacts, the co-benefits and trade-offs of actions. If these are not accounted for, legislation may be sub-optimal, and perhaps contradictory in places. An example of the latter concerns the promotion of biomass burning for its claimed climate mitigation impacts, despite its high emission of particulate matter (Holland, 2018).

A particularly extensive evaluation of co-benefits and trade-offs linked to climate policies is provided by Smith et al (2016). The systematic accounting of ancillary impacts of environmental policy developed in that work provides a template for similar analysis in other policy areas.

Similarly, referring back to Figure 4-15, there are strong links with the Birds[30] and Habitats Directives[31], given the extent of exceedance of critical loads for eutrophication in Natura 2000 Sites. This further underlines the need for impacts to ecosystems to be accounted for in the analysis, and for results, whether in a monetised form or not, to be taken forward to the subsequent decision-making process.

Further links may be drawn with social legislation, for example on equity, given the potential for those who are least well off to live and work in the most polluted environments.

4.2.5 Better Regulation agenda

The objectives of the Better Regulation agenda[32], accompanied by the Better Regulation Guidelines and Toolbox[33], are to ensure that:

- decision-making is open and transparent
- citizens and stakeholders can contribute throughout the policy and law-making process
- EU actions are based on evidence and understanding of the impacts
- regulatory burdens on businesses, citizens or public administrations are kept to a minimum

The quantification of impacts and subsequent monetisation can address several of these issues. It can help to open out the rationale for decision making, and via the use of monetary valuation, directly provide information on the weights attached to different types of effect. These economic weights are based either on market prices or on an expression of public preference for the allocation of resource. Providing this information in an open way gives stakeholders the opportunity to comment on proposals from a more informed position than would otherwise be possible. The analysis also demonstrates the evidence base for taking action. Basing results on the outcomes of cost-benefit analysis also provides some guide (recognising that they will take into account additional factors that may be outside of the analysis) for policy makers towards minimisation of the regulatory burden relative to the benefits of agreed actions.

This assumes, however, that the methods used for analysis are unbiased and provide a realistic and honest interpretation of available research. This in turn places an onus on analysts to be clear as to why they have (e.g.) used particular models for dispersion calculations, and adopted certain response functions and valuations. It places a further onus on analysts to report on uncertainty and to provide guidance on what uncertainties are covered and not covered in that assessment.

4.2.6 Consequences for national damage cost quantification

The issues raised above highlight a number of principles that should be considered for national damage cost quantification:

[30] http://ec.europa.eu/environment/nature/legislation/birdsdirective/index_en.htm

[31] http://ec.europa.eu/environment/nature/legislation/habitatsdirective/

[32] https://ec.europa.eu/info/law/law-making-process/planning-and-proposing-law/better-regulation-why-and-how_en

[33] https://ec.europa.eu/info/law/law-making-process/planning-and-proposing-law/better-regulation-why-and-how/better-regulation-guidelines-and-toolbox_en

1. Analysis should be consistent with the Commission's Better Regulation guidance.
2. Best practice would be to provide a complete quantification of known pollutant damage, across different media, over extended distances not limited to national boundaries, and for impacts to health, ecosystems and the built environment.
3. Analysis should aim to be transparent in order that stakeholders can understand the quality of models and data used, the assumptions made, and uncertainties present, including the omission of any categories of impact.
4. Links to other policies, for example on climate, via the presence of co-benefits or trade-offs ought to be identified and brought into the quantification if possible.
5. It is advised that areas where a precautionary approach may be warranted are identified, for example in relation to possible irreversible damage associated with ecosystem eutrophication. Damage costs should still be quantified to inform the policy process, but the decision making process must transparently take account of such damage. This does not necessarily presume in favour of the precautionary principle.

4.3 The impact pathway and unit damage cost approaches

4.3.1 Overview of the IPA and damage cost approaches

The impact pathway approach (IPA) has been widely adopted for the quantification of health and environmental damage associated with the release of pollutants since its development in the EC funded ExternE (Externalities of Energy) study of the 1990s and early 2000s (Berry, Holland, Watkiss et al, 1995). Early applications focused on air pollution, though its use has increased significantly in the last decade in the field of chemicals as a result of the REACH Regulation.

Quantification following the IPA typically proceeds through the stages shown in Figure 4-16:

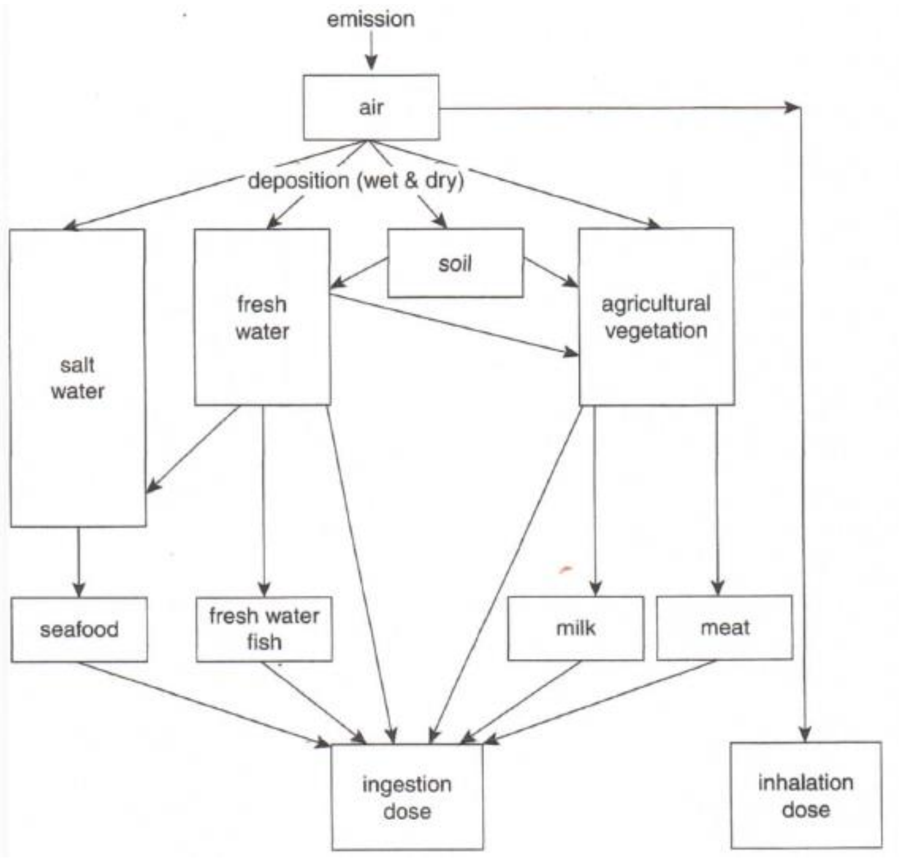
Figure 4-16: Illustration of the impact pathway approach (IPA)

1. Quantification of emissions
2. Modelling of pollutant dispersion and chemistry
3. Exposure of people and sensitive environmental receptors
4. Quantification of impacts, using concentration-response functions
5. Valuation of quantified impacts
6. Review of uncertainties, including additional factors

The steps of the impact pathway are elaborated in more detail below. The method can be applied to individual installations, and by doing so account for variation in factors that will influence dispersion, such as stack height, flue gas temperature, emission exit velocity, local topography and local meteorology, and factors that influence exposure, notably the relationship between the location of a plant and the distribution of sensitive receptors (people, ecosystems, materials) around it. An early finding from the ExternE work was that it is necessary to model dispersion over long distances, for large power stations in the order of several hundred or more than 1,000 km, to capture impacts more or less completely (Berry, Holland, Watkiss et al, 1995, p.35). The method can also be used at the national level, as used by the European Environment Agency for quantification of national damage costs per unit emission, and some national governments (e.g. Denmark and UK). The averaging implicit in any national level approach inevitably introduces greater uncertainty when applying the resulting damage costs to a specific facility.

The linear structure of the impact pathway above is appropriate to the major air pollutants (PM, ozone, NH_3, NOx and SO_2) where damage to health is linked via response functions only to inhalation, and damage to ecosystems to either the concentration or deposition of pollutants. For application to trace pollutants such as dioxins or toxic metals, where response is linked to dose via ingestion to a much greater extent than inhalation, the impact pathway becomes more complex (Figure 4-17) (Rabl et al, 2014).

Figure 4-17: Impact pathway for toxic metals, dioxins and other pollutants for which relevant exposure occurs via both ingestion as well as inhalation. Source: Rabl et al (2014).



Quantification using this expanded impact pathway requires knowledge of transfer factors through the environment, for example from air to freshwater to fish and to human ingestion. These factors are available and have been used for impact pathway modelling, for example by Joseph Spadaro using the RiskPoll model (see Annex 3 of the report on damage from industrial installations by the European Environment Agency, 2014).

The two major constraints on use of the IPA concern:

1. The level of knowledge and effort, and access to models, that are required to undertake the modelling.
2. Limited availability of data to permit quantification of some important impacts.

To address the first point, a simplified approach based on unit damage costs (expressed as, e.g., €/tonne emission) has been developed, enabling estimation of damage based only on knowledge of emissions and the unit damage costs. It has, notably, been used by the European Environment Agency to quantify the damage associated with plant reporting to the European Pollutant Release and Transfer Register (E-PRTR).

It is sometimes inferred that the IPA and the use of unit damage costs are distinct approaches with little in common. Instead, they are closely related, with unit damage costs such as those reported by the European Environment Agency, being generated using the IPA. The difference between them is that the IPA can be adapted from the outset to the specific characteristics of plant that it is being used to model (stack height, location relative to sensitive receptors, etc.), whereas unit damage costs are an approximation, though factors can be applied to make them more specific to the case under analysis (see, for example, Annex 4 of the EEA report (2014), which concerns Sectoral Adjustment).

The second issue, limited availability of data, is a reflection of the current state of science. Without further research, this can only be dealt with through the narrative provided with results. This, in turn, needs to be written in a way that can be communicated through the decision making process, although it is far easier to carry numbers through the process, especially those expressed in economic terms, than a narrative description. It should be recognised that limited data availability is not always a serious problem: for fine particles, for example, there are certainly gaps in knowledge, e.g. in relation to the quantification of effects on development in the young and instigation of dementia in the elderly. However, effects that can be quantified with a very good level of confidence, particularly on mortality, are generally sufficient to justify action being taken. For some other pollutants, notably trace pollutants where epidemiological evidence may be far from complete, the limitation may be more problematic. The scope for significant underestimation of damage costs for trace metals via the omission of impacts is highlighted in the papers by Nedellec and Rabl discussed below (2016a, b).

Since the mid-1990s, when the IPA became operational as a result of advances in computing power combined with emergence of data particularly on concentration-response functions, there have been no pollutant damage costs developed and published based on methods other than the IPA to the knowledge of the present authors. Some previous work, for example by Hohmeyer (1989), adopted a cruder ‘top-down’ analysis that allocated some fraction of national impacts to emissions of various kinds. However, the authors of similar papers recognised the need for an approach linked more closely to the science that was emerging at the time, and moved over to use of the IPA, or results derived from the IPA, when they became available. Some other early work used estimates of willingness to pay to live in cleaner environments, using the hedonic pricing method with value of environmental quality reflected through property price. Smith and Huang (1995) performed a meta-analysis of 37 studies that demonstrated a clear relationship between air pollution and property price. However, these studies were falling out of favour even before the advent of the IPA, with most of the studies reviewed by Smith and Huang being published prior to 1980. The reason that this type of work is no longer performed specifically for air pollution is likely that the IPA is regarded as a superior alternative, as it is explicit about impact and hence what needs to be valued.

4.3.2 Overview of methods for each stage of the analysis

Before considering national damage cost estimates from various sources, this section provides information on factors relevant to each stage of the analysis, addressing the following points:

1. The purpose/outputs of each step
2. The process/calculation undertaken
3. The tools, methods, data inputs and sources used in each step
4. Decisions that need to be made when developing the methods
5. Potential sources of divergence in analysis
6. What aspects can be made ‘country-specific’, and what impact will this have?

Discussion here focuses on the quantification of impacts associated with exposure to fine particles and ozone, linked to emissions of NH_3, NOx, $PM_{2.5}$, SO_2 and VOCs, following the general structure of the linear impact pathway shown in Figure 4-16 (the same applies to any pollutant whose primary pathways are linked to inhalation, so will include some metals and trace organics, but not all). The discussion focuses on these pollutants as the evidence base linking exposure to health outcomes is more developed for these pollutants than for others. Some discussion is also provided on data needs, etc. for implementation of the more complex pathway involving cross-media transfer shown in Figure 4-17.

Quantification of emissions

Quantification of emissions provides the foundation of both the IPA and the application of damage costs per unit emission. It identifies the quantity of pollution for which the associated damages are being estimated. From a bottom-up perspective it may be performed through knowledge of:

- Emission concentrations in flue gases and waste waters
- The content of metals, etc. in input materials relative to that contained in product

Emissions data for individual plant are available from the European Pollutant Release and Transfer Register (E-PRTR), and these data, to the extent that they are available, can be used as the baseline for an ex-post analysis. It is noted that the validity of the reported data can be an issue given the methods for measuring, estimating and calculating emissions on the facility level.

The same data can be aggregated to the national level. A limitation arises because of the reporting thresholds used for the E-PRTR by most countries, which may lead to a significant underestimation of the benefits of reducing trace pollutants, such as toxic metals: the fact that many plants report no emission of numerous substances does not mean they are not being emitted, but that the quantities concerned are below the reporting threshold. Other data sources include national emission inventories [34] the returns made to the EEA and UNECE to demonstrate performance against legislation such as the National Emission Ceilings Directive and the Convention on Long Range Transboundary Air Pollution. However, disaggregation of some of the data provided by these sources may prove difficult when seeking to assess the effect of the IED.

The key decisions that need to be made at this stage relate to:

1. Whether to quantify the change in emissions from the IED bottom-up using data on individual plant, or top-down using national data. The former is likely to be more accurate, noting the issue of attribution raised below, though will be more time consuming.
2. The number of pollutants to be considered in the analysis.
3. The number of years over which performance before and after the effects of the IED are experienced. Emissions can vary significantly from year to year for reasons unrelated to emission controls, for example, from the breakdown of machinery or refurbishment or further development of a site.

There is an important issue of attribution of emission changes to specific legislation that must be considered. This is especially important at the present time given the joint effects of the IED and climate legislation. One example concerns the promotion of industrial burning of biomass for power and heat generation under climate legislation, that may prolong the lifetime of some large coal fired power stations and change the pollution generated by a site (Fern, 2018).

The quantification of emissions is entirely country-specific, as it needs to reflect the mix of activities and fuels that are present. These vary significantly, with some differences having grown in recent years through the demise of coal in some regions, and increased coal use in others, such as the Balkan region.

Dispersion of pollutants

The second stage of the impact pathway involves the use of detailed and complex pollutant dispersion models to assess:

- The spread of emitted pollutants around the release site or sites. For the purpose of developing national damage cost estimates it is possible to scale the modelling up to the national level.

[34] Examples include the Dutch Pollutant Release and Transfer Register (http://www.emissieregistratie.nl/erpubliek/bumper.en.aspx), the Swedish Pollutant Release and Transfer Register (http://utslappisiffror.naturvardsverket.se/en/) and the UK's National Atmospheric Emission Inventory (NAEI: http://naei.beis.gov.uk/).

- Chemical transformations of pollutants leading to the formation of secondary species, some of which (e.g. secondary inorganic and organic particles, or ozone) are harmful to both human health and ecosystems.

Reference is made to the response functions identified for impact assessment, to ensure that the dispersion modelling provides data on the concentration and deposition of pollutants in the correct metrics, and averaged over the correct time period.

Dispersion models take account of a number of parameters in addition to the quantity of pollution released, including the design of the pollutant release system (stack height, diameter, flue gas temperature), meteorological conditions and wind direction. When dealing with reactive pollutants such as NH_3, NOx, SO_2 and VOCs it is essential that the models account for chemical reactions.

Many of the tools developed for modelling dispersion from industrial facilities are not appropriate for the purpose of damage cost assessment. These models often operate over only short ranges (to a few tens of km) and many do not incorporate pollutant chemistry. These have been developed primarily for assessment of compliance with air quality limit values. Range can be limited because the zone likely to be affected by the largest increment in pollution levels associated with an industrial site will be relatively close by. These models are not suitable for quantification of the benefits of the IED because they will omit a major part of the effects of emitted pollutants from industrial sites.

There is a view that national modelling, within one's own national boundaries, will provide a materially more robust estimate of damage or the benefits of emissions controls, than modelling over a coarser scale across Europe. However, this is not necessarily the case. Emissions from tall stacks prevalent across many of the sectors to which the IED applies, spread over very large areas, so the part of damage or benefit that arises within one's own national territory may be small, in many cases less than half of the total estimate. It is possible to combine the results of models operating at different scales, but this generates some risk of systematic over- or under-estimation of effects, for example by double counting of the deposition of pollutants.

There are, however, several models available, developed across Europe, for carrying out the necessary calculations, including the full European scale and pollutant chemistry. A recent model comparison exercise provides some overview of the models CAMx, CHIMERE, CMAQ, EMEP, LOTOS-EUROS, MINNI and RCG (Bessagnet et al, 2016). These models have been developed over a number of years by national institutes (e.g. INERIS for CHIMERE, USEPA for CMAQ, Norwegian Meteorological Institute for EMEP, ENEA for MINNI), major consultancies (Ramboll-Environ for CAMx, TNO for LOTOS-EUROS) and universities (Freie Universität Berlin for RCG).

Given the influence of meteorology on pollutant dispersion, results from these models are generally averaged over a number of years. The EMEP model, for example, provides analysis over 5 meteorological years, including extremes (e.g. hot and dry years vs cool and wet years).

The ultimate outputs of this step are changes in pollutant concentrations, associated with the change I underlying emission.

Exposure of people and sensitive environmental receptors

'Exposure' is expressed simply as the extent to which the population and other receptors are exposed to pollution. The next step therefore involves combining the pollutant dispersion model output with data on the distribution of the population and other sensitive receptors across Europe. For human health, for example, this will lead to a summary measure combining concentration and population, typically in the form of average population weighted concentration, calculated as population x concentration in each grid cell, summed to the national level, then divided by total population.

Reference is again made to the impacts for which response functions are available, to ensure that exposure data are appropriate to the functions that will subsequently be used.

A full assessment requires spatial data regarding the distribution of:

- People, in different age groups and with differing levels of disease
- Agricultural crops, by species
- Forests, by species

- Natural ecosystems, by type, and possibly also recognising areas under statutory controls (e.g. the Natura 2000 sites)
- Materials used for buildings and vehicles (these are the two most significant uses of materials so far as the response functions are concerned). The material inventories should ideally be broken down to different types of material including limestone, sandstone, mortar, concrete, steel, galvanised steel and rubber.

Where resources are limited, assessment should focus on human health impacts as these account for roughly 90% of quantified monetised damage (see Holland, 2014a). If analysis is restricted, it is important to recognise those impacts omitted from the assessment and carry them through to the discussion of results.

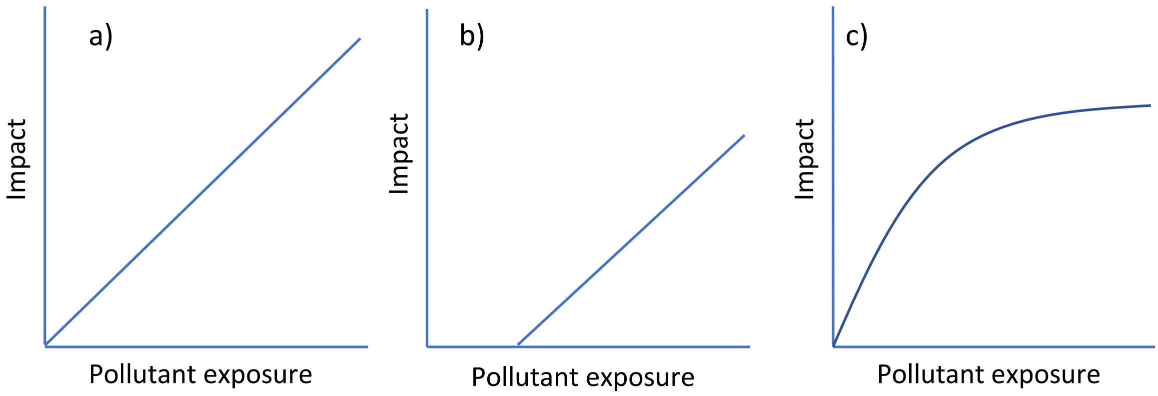
These data have already been collated for impact quantification, for example in the AlphaRiskpoll model that has been used for benefits assessment at both European and national levels (Holland 2014a).

For assessment of the benefits of the control of pollutants under the IED over any significant period of time (say 10 years or more) it is appropriate to consider future changes in population, both in terms of the overall number of people concerned and the age and health profile of the population. Data are available from Eurostat, the UN World Population Prospects and WHO. In theory, similar account should be made of changes in ecosystems, agricultural production, etc. However, given that damage to these receptors adds only a small amount to total estimates of damage or benefit, and the precise extent of change is difficult to forecast, this has so far been ignored.

Quantification of impacts using concentration-response functions

Response functions link a measure of exposure, typically as population weighted concentration (for health, crops and forests) or deposition (for materials and ecosystems) with impacts. Concentration-response functions[35] can take a number of forms, some of which are shown in Figure 4-18. For quantification of health impacts in Europe it is generally agreed that (a) linear with no threshold is appropriate, though (b) and (c) from the figure also feature in the literature.

Figure 4-18: Illustrative shapes for concentration-response functions. a) linear, no threshold. b) linear, with threshold. c) non-linear.



WHO-Europe recommended a set of response functions for European analysis of health impacts through the HRAPIE (Health Risks of Air Pollutants in Europe) study (World Health Organization, 2013), through debate with a large number of experts from Europe and North America. HRAPIE was concerned with the effects of exposure to ozone, fine particles and NO_2, though this feeds through to damage costs for a wider range of pollutants (NH_3, NOx, primary $PM_{2.5}$, SO_2 and VOCs) as a result of chemical reactions in the atmosphere. Ecosystem impacts and damage to materials can be quantified using functions recommended by the groups operating via the Working Group on Effects under the UN ECE Convention on Transboundary Air Pollution. Although the above represent widely adopted approaches,

[35] Reference is made in the literature also dose-response, or exposure-response functions. The three terms tend to be used interchangeably when linking some quantity of pollution with effects, though they differ in definition.

the set of impact pathways included in analysis can vary, as illustrated in the examples set out in Section 4.5.

Guidance on implementation of the HRAPIE functions, including sources of data on background incidence of disease, was provided by Holland (2014b). Some of the response functions, particularly for mortality, require the use of life tables rather than a simple multiplication of population affected by concentration and response function. IOMLIFET, an online tool, has been developed for this purpose by the Institute of Occupational Medicine in Edinburgh [36]. However, once initial runs have been performed it is possible to extrapolate life table results to different scenarios.

For ecosystems and materials there is little divergence in approach to quantification across Europe. However, health impact assessment raises several potential sources of divergence in analysis:

- The pollutants for which quantification should be attempted
- The extrapolation of response functions down to very low concentrations
- The shape of response functions
- The range of response functions included in the analysis (i.e. the coverage of health impacts)
- The use of national studies rather than the wider international literature
- For trace pollutants especially, identification of response functions reflecting the disciplines of risk assessment or economic assessment.

Considering first the question of the pollutants for which quantification should be attempted. A common assumption is that a response function defined against exposure to a particular pollutant (e.g. $PM_{2.5}$ or NO_2) reflects the impacts of that pollutant specifically. This is not necessarily the case, given that many pollutants share a common source (particularly the combustion of fossil fuels) and in the underlying epidemiological studies it is often impossible to isolate the impact of an individual pollutant. In reality, a single response function could reflect the health impact of the main pollutant mixed with other pollutants. For that reason, where more than one pollutant is assessed using damage cost functions, there is a risk of double counting effects because some of the impacts associated with reduction in emissions of one pollutant can already be captured in the assessment of another pollutant. Response functions can therefore be interpreted as showing:

- The effects of a specific pollutant
- The effects of a pollutant mix, characterised for the purposes of quantification by a specific pollutant, but with results representing the effects of exposure to the full mix.

The results of response functions for PM and for ozone can be added together without risk of double counting, as demonstrated by the HRAPIE study. On the other hand, it is recognised that there will be some overlap in the results of functions quantifying against PM and NO_2 exposures because the two pollutants share a common source. Within analysis at the European scale so far this has not affected results as the NO_2 functions have not been used, given concerns about the ability of the dispersion models to operate well at scales appropriate to assessment of NO_2 exposure. However, it is an issue to be aware of for the future, as dispersion modelling for NO_2 becomes available.

Standard scientific practice is not to extrapolate data beyond the range of observations. In past times, when epidemiological studies were carried out typically in cities that were known for poor air quality this raised significant questions for the analysis. However, over time, these studies have been carried out across a much wider range of locations, including those with low pollutant levels, and so a more informed position can be taken. There is general (if not always unanimous) agreement on the following:

- No threshold for effects of PM, NO_2 and trace metals and organic pollutants [37]
- No identified threshold for ozone, but (from HRAPIE) quantification above a concentration of 35 ppb will provide a more reliable result than quantification at lower concentrations. The figure

[36] https://www.iom-world.org/news-events/news/2013/iomlifet/

[37] HRAPIE recommends use of a 20 ug.m^{-3} cut point for analysis of the effects of chronic exposure to NO_2 on mortality, though not for other NO_2 endpoints, reflecting limited evidence on the effect of exposures below this level. More recent review (e.g. by COMEAP in the UK: https://www.gov.uk/government/publications/nitrogen-dioxide-effects-on-mortality) reports on the publication of studies that include more observations at low concentrations, and find no evidence for a threshold.

of 35 ppb is referred to as a 'cut-point' for analysis (i.e. a level below which there may still be effects) to distinguish it from a 'threshold' (a level below which there are no effects). However, HRAPIE suggests to quantify using both the cut-point of 35 ppb and quantification without use of a cut-point.

With respect to analysis in the context of the IED, uncertainty about the validity of extrapolation to zero for the PM response functions that tend to dominate analysis is of little relevance, given the non-zero background pollutant levels that are present if emissions from IED regulated facilities are ignored. Modelling tools have been developed that account for the cut-point recommended for application with ozone assessments.

Whilst the HRAPIE study recommends the use of functions that are linear with no threshold for quantification of PM effects, the Global Burden of Disease study (IHME, 2018)[38] has taken a different approach, with quantification above a threshold, and response levelling off at higher concentrations (a combination of Figure 4-18b and Figure 4-18c). The Integrated Exposure Response functions used by GBD reflect concern that response will saturate at higher concentrations (consistent with data on exposure to particles from other sources). However, HRAPIE concluded that the linear function recommended by the study was preferable for analysis at concentrations typical of the European region. They further concluded that quantification should be based against all-cause mortality functions rather than cause-specific functions (lung cancer, respiratory disease, circulatory disease), noting unexplained variability in cause specific mortality rates between countries.

Some analysts prefer to use response functions derived from epidemiological studies carried out in their own country, regarding this as likely to provide a more accurate result by implicitly accounting for the state of health within the country concerned. However, this may not work as intended: extensive variability has been observed in studies within a single country, and it is generally considered better practice to draw on as large a sample of data as is reasonable. The use of evidence from a single country is also inconsistent with the need to quantify transboundary impacts.

The final issue identified above concerned the quantification of impacts of trace pollutants (toxic metals, organics), whether to follow typical practice in chemical risk assessment or not. Chemical risk assessment often focuses analysis on a narrow range of effects, perhaps just one of several that are plausible, to answer the question of whether or not a risk is present and broadly how large it is. Selected effects will be those for which most data exists, and which tend to be considered serious (e.g. development of cancer, loss of IQ). For economic assessment, where it is necessary to compare costs and benefits, results will be biased by a failure to quantify effects. If those selected for quantification are likely to dominate the analysis this bias will be small, but Nedellec and Rabl (2016a) demonstrate that this is not always the case, drawing on analysis of arsenic, cadmium, lead and mercury. Countering this, and reflecting the discussion above concerning the difficulties of separating out impacts of different pollutants emitted from the same source, it is likely that some of the impact of exposure to trace metals will be captured by response functions addressing long-term (chronic) exposure to particles. With that in mind it seems appropriate to limit the quantification of the impacts of several toxic metals to types of effect that are not quantified for particles in aggregate estimates of the benefits of the IED (i.e. where effects across multiple pollutants are captured). It would be appropriate, however, to undertake a more complete quantification specifically for the metals if the objectives of analysis required a deeper understanding of their effects. Mercury is an important exception to this: it's volatility means that its impacts will not be captured in the response functions for PM. Mercury suspended on particles is considered negligible because the majority of emissions will be in gaseous form. This discussion also does not apply to quantification of the impacts of most organic pollutants for which combustion is not a key source.

The output of this step are estimates of the change in health outcomes (e.g. change in hospital admissions) associated with the initial change in emissions.

Valuation of impacts

Impact valuation is simply performed by multiplying quantified impacts by a set of unit values (e.g. expressed as € per hospital admission) and aggregating over the full set of impacts. The output of the analysis is an expression of public preference for the allocation of resource to avoid damage. It does

[38] http://www.healthdata.org/infographic/global-burden-air-pollution,

not attempt to define the 'intrinsic worth' of human life, healthy ecosystems, etc., though it does extend beyond quantification of marketed goods such as crops, timber or labour.

The valuation process brings together:

- Direct economic losses through increased healthcare costs, material damage, crop losses, etc.
- Lost production through worker illness
- Lost utility via pain and suffering when unwell, loss of ecological quality, etc.[39]

As noted elsewhere, health impacts dominate when valuation is applied. An earlier approach for health valuation, the human capital approach (HCA), focused only on the first two points in the above list. However, it is recognised that this is incomplete and that utility losses should be accounted for, and use of the HCA has largely vanished from the European literature on pollution controls. Valuation accounting for lost utility is based on elicitation of individual 'willingness to pay' (WTP) to avoid, or 'willingness to accept' (WTA) either using market data, or more commonly, using questionnaires to assess the value that individuals place on reducing or increasing the risk of illness. Use of market data is largely restricted to mortality, where experts in the US have used 'wage-risk' studies for many years, deriving a 'value of statistical life' (VSL) by assessing how wage rates change relative to the risk faced by people in different occupations requiring similar skills. In Europe the contingent valuation approach is preferred (also by OECD (2012)). Calculations of contingent valuation are broadly identical to the wage risk studies with WTP equated against the change in risk.

In European policy analysis quantification of mortality damage has used two approaches, one based on valuation of estimated deaths using the VSL (as discussed above), and the other based on valuation of lost life expectancy using the Value of a Life Year (VOLY). Opinion is split on which is preferable, hence both are analysed. Examples of sources for such values, and the range of values applied can be found in Section 4.5.

There are several areas where analysis by different groups may diverge, the most important of which largely concern health valuation:

- Approach used for mortality valuation: VSL or VOLY. This can make a difference of a factor 3 or more to final estimates of damage or benefit.
- The estimated value for mortality (i.e. the € per VOLY or € per VSL), again with a factor 3 or 4 variation in estimates from the main valuation studies used in the literature.
- Factoring forecast changes in economic growth to the analysis, with consequent effects on WTP.
- The use of values averaged across the EU with respect to income, or country-specific values.
- Interpretation of economic assessments of ecosystem damage. In particular, whether they provide a sound basis for comparison of costs and benefits (no consideration is made here of the use of the human capital approach as it has long been rejected in Europe).

Review of uncertainties and additional factors

The key question faced in assessment of uncertainties in cost-benefit analysis concerns whether any of the uncertainties that are present are likely to alter the conclusions reached on the balance of costs and benefits, in other words whether the finding that benefits exceed costs, or vice versa, could change.

Uncertainty analysis can draw on a variety of methods:

- Use of Monte Carlo techniques to bring together data on variation in each of the many quantified elements of the analysis.
- Sensitivity analysis addressing methodological assumptions. The most important of these seem likely from the discussion above to concern mortality valuation and the inclusion of NO_2 effects.

[39] There is some debate as to whether these three categories of value are additive when lost utility is valued using contingent valuation, as respondents could factor in lost earnings and health care costs to their response. It is considered here that it is appropriate to combine the estimates given the social welfare systems that are prevalent across Europe.

- Qualitative review of the biases that affect the analysis, identifying for each the likely direction of bias.

A framework for bringing together this information on uncertainty in a concise manner was developed under the EC4MACS study (Holland, 2013).

The key decision that needs to be made concerns the number of uncertainties that need to be accounted for in the analysis. Uncertainties relating to mortality quantification will be important in all cases, given that it dominates the analysis.

4.4 Review of the EEA damage costs

4.4.1 NH_3, NOx, $PM_{2.5}$, PM_{10}, SO_2 and VOCs

The unit damage costs produced for the EEA (2014) for emissions of NH_3, NOx, $PM_{2.5}$, PM_{10}, SO_2 and VOCs are provided at the national level already and hence are country specific. In this context, 'the national level' concerns the impact of emission of one tonne of pollutant from a particular country, wherever the impacts occur. The EEA damage costs provide estimates of damage per tonne averaged across all sources of each pollutant in a country (i.e. industry, transport, domestic, agriculture, etc., noting that Annex 4 of the EEA report describes methods for adjusting values so that they are better representative of industrial emissions). The data sources used are as follows:

- Pollutant dispersion and chemistry: EMEP model [40] transfer matrices, as used in the GAINS model [41] to inform development of EU's Thematic Strategy on Air Pollution.
- Population data: Eurostat
- Response functions: HRAPIE (Health Response to Air Pollutants in Europe) study carried out by WHO-Europe on behalf of the European Commission and involving numerous experts from European academic and health institutes, and also some North American experts. Results also included analysis of damage of ozone to crops and of acidification to building materials. Analysis here drew on earlier work, though used accepted European response functions. Original work on these receptors was not considered necessary given that the overall damage costs are dominated by health impacts.
- Valuation data: As adopted for cost-benefit analysis of the Thematic Strategy on Air Pollution (Holland, 2014a) with average EU values applied in all Member States.

The EEA damage costs for each pollutant cover impacts on human health from exposure to fine particles (both primary and secondary species) and ozone. HRAPIE also recommended functions to account for effects of exposure to NO_2, though these were not applied in the damage cost estimation given concerns about the ability of the pollutant modelling to provide an estimate of exposure consistent with that used to characterise the exposure-response functions from the epidemiological literature. The European Commission has since funded further work to provide an improved modelling framework in this area, though this has yet to be applied for damage cost estimation (VITO, 2017). A further concern on this point is that the HRAPIE recommendations on NO_2 may now be outdated, given the publication of a significant volume of new research on the pollutant since HRAPIE was finalised (as reviewed by COMEAP, 2018). The principal findings of the COMEAP assessment were that there was no evidence for a threshold for effects associated with NO_2. However, it was also concluded that response is likely to be weaker than indicated by HRAPIE per unit of exposure. Overall, impacts would be higher at low concentrations, but lower at higher concentrations. Significant uncertainty was reported in attribution of impacts to NO_2 and other related burdens of traffic. Some of the authors questioned a role for NO_2, per se, rather than potential correlated variables, though there was agreement that the functions adopted for PM did not describe the full burden of polluting activities.

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[40] http://www.emep.int/mscw/mscw_models.html

[41] https://ec.europa.eu/clima/sites/clima/files/strategies/analysis/models/docs/gains_en.pdf , http://www.iiasa.ac.at/web/home/research/researchPrograms/air/Program-Overview.en.html

A limitation of the analysis of VOCs arises because the assessment includes only impacts associated with exposure to ozone and secondary organic aerosols. Other impacts of VOCs, such as direct health effects of exposure, were not included. A more disaggregated analysis of VOCs by species would be possible if it was needed: this could take into account any direct health and ecological impacts of a substance and variation in photochemical oxidant creation potential (POCP), organic aerosol formation potential, etc.

A possible omission in quantification of effects of NO_2 also concerns impacts of acidification and, more particularly, eutrophication on ecosystems. There has been some progress made in this area via the ECLAIRE study (Holland et al, 2015a,b) funded under FP7. This work provided damage costs for forests, crops and ecosystems. Including these impacts would not substantially change the estimates, given the dominance of health impacts. However, further debate on this issue would be useful, particularly in relation to ecosystem impacts, given the extent of exceedance of the critical load for eutrophication which opens a possibility for significant ecological harm.

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

1. The first set is included the BREF-13 on Economics and Cross Media Effects published in 2006 (European Commission, 2006)
2. The second set was included in the EEA report 'Revealing the costs of air pollution from industrial facilities in Europe', published in 2011 (EEA, 2011)
3. The third set was included in the EEA report 'An updated assessment of the damage cost estimates to health and the environment caused by pollutants emitted to air from Europe's largest industrial facilities' published in 2014 (EEA, 2014)

The first two sets of results are now outdated and should not be used. Each subsequent set of damage costs has used updated response functions, pollutant transfer matrices and other data, drawing on best practice and consensus across European experts, as illustrated by the HRAPIE study carried out for the European Commission by WHO-Europe.

4.4.2 Trace pollutants: Toxic metals and organics

The methods used for quantification of the damage costs of these pollutants are described by Joseph Spadaro in Annex 3 of the EEA (2014) report, complete with transfer and bioaccumulation factors. For most of these pollutants costs are provided at national level, whilst lead and mercury are only provided as European averages. The rationale for this is that the response functions for lead and mercury deal with total ingested dose (and subsequent effects on IQ), and that this dose will be dominated by intake from food. Given the international trade in food (particularly fish, in the case of mercury) any assumption of purely local production was considered likely to generate an unreliable result.

There is limited information from outside the EU, or within individual Member States on damage costs for the trace pollutants considered in this section.

Proceeding through the stages of the impact pathway, the EEA damage estimates used the following:

- Pollutant dispersion and chemistry: Uniform World Model (Spadaro, J. V. and Rabl, A , 2004; 2008).
- Environmental transfer factors: USEPA (2005), and other sources listed in Annex 3 to the EEA report.
- Population data: Eurostat
- Response functions: Various sources, including USEPA, WHO, and the EC funded NEEDS and Methodex studies, as listed in the EEA report.
- Valuation data: As adopted for cost-benefit analysis of the Thematic Strategy on Air Pollution (EEA, 2011) with average EU values applied in all Member States. For IQ loss, loss of lifetime earnings drew on analysis by Rabl and Spadaro from an earlier paper (2014).

The use of the Uniform World Model may appear simplistic compared to the large pollutant dispersion and chemistry models available in Europe such as EMEP. However, cross-checking of results indicates that any difference in results is certainly within the bounds that may be associated with other parameters such as the response functions used in the analysis. The response functions applied for the pollutants in question are typical of those used elsewhere (for example in assessment of lead and mercury in the context of Restriction dossiers produced under the REACH Regulation). Similarly, transfer and bioaccumulation factors and valuation data are typical of wider practice.

The major question around the quantification of damage costs for these pollutants concerns the range of impacts that should be considered. Spadaro's work for the EEA followed generally accepted practice and quantified against only a limited number of effects for which there is general agreement (such as effects of lead and mercury on IQ loss, or cancers linked to exposure to arsenic, dioxins, PAHs etc.). Later work in France by Nedellec and Rabl (2016a,b, also used above in Section 2.2.2.9) demonstrates that extending analysis to other plausible endpoints can greatly magnify damage cost estimates: their estimates of lead and mercury damage increased by a factor of 10 when effects on mortality were included alongside IQ loss. Similar large increases were found for arsenic and cadmium.

This clearly raises the question of which effects should be included in the analysis, an issue also reviewed by Dubourg (2018). From a chemicals risk-analysis perspective there is a tendency to focus on one or a limited number of effects that:

- Are individually serious (e.g. mortality or IQ loss)
- Are the subject of a number of research papers, that
- Provide consistent indication of impact

A problem with mercury, for example, is that intake is largely via consumption of oily fish. There is then a tension between the negative impact of ingesting mercury on the circulatory system, with benefits from consuming the fish. This tension was considered by Nedellec and Rabl (2016) to explain the inconsistency in results.

The purpose of chemicals risk analysis is typically to say whether or not a risk of some kind is present, and whether it is significant. From this perspective it is appropriate to focus on one or more effects that are characterised with the highest confidence. From an economic perspective, however, a broader analysis may be appropriate to quantify the major part of damage for input to CBA. The publication of the Nedellec and Rabl papers in a well-regarded peer reviewed journal indicates at the very least that the reviewers considered that a good case for extension of the analysis was merited. However, in the two years since publication there has not been a rush by analysts to adopt the same assumptions on causality, though they are used above. This is an area where further expert review would be useful, with a view to recommending impacts that can be brought forward to CBA. Discussion would be needed between experts from the different disciplines so that concerns about uncertainty could be properly integrated to the analysis.

4.5 Making damage costs country-specific

This section includes consideration of applications of benefits analysis and commentaries on methods by the following countries or bodies:

- Quantification of damage costs for the Danish Government (Andersen & Brandt, no date).
- Quantification of damage costs by the Department for Environment, Food and Rural Affairs, UK (DEFRA, 2013; 2015a,b)
- Analysis by the United States Environmental Protection Agency (USEPA) of the benefits of controlling hazardous air pollutants from power stations (US EPA, 2016) and a prospective analysis of the benefits of the US Clean Air Act from 1990 to 2020 (US EPA, 2011).
- Holland (2017) for Frank Bold, an NGO based in the Czech Republic, with the paper being reproduced on the websites of a number of other European NGOs, including the Health Environment Alliance. The paper seeks to assist NGOs in understanding the application of damage costs with respect to applications for derogation under the IED.
- Quantification of unit damage costs for the European Environment Agency (EEA) (Holland, 2017).

Note that discussion of making unit damage costs sector specific is given in the next section.

4.5.1 Pollutants considered

All five of the bodies considered here cover a range of pollutants. The most extensive assessment in this regard is by the EEA (and by association, Holland (2017) for Frank Bold, which adopted the EEA method, though includes further guidance).

Table 4-1: Coverage of pollutants by different organisations

	Denmark	UK	USEPA	Frank Bold	EEA
NH_3	✓ via $PM_{2.5}$	✓ via $PM_{2.5}$	✓ via $PM_{2.5}$	✓ via $PM_{2.5}$	✓ via $PM_{2.5}$
NO_x	✓ via $PM_{2.5}$, O_3, NO_2	✓ via $PM_{2.5}$, O_3, NO_2	✓ via $PM_{2.5}$, O_3	✓ via $PM_{2.5}$, O_3	✓ via $PM_{2.5}$, O_3
$PM_{2.5}$	✓	✓	✓	✓	✓
PM_{10}		✓	✓	✓	✓
SO_2	✓ via $PM_{2.5}$	✓ via $PM_{2.5}$	✓ via $PM_{2.5}$	✓ via $PM_{2.5}$	✓ via $PM_{2.5}$
VOCs		(✓) via O_3	(✓) via $PM_{2.5}$, O_3	(✓) via $PM_{2.5}$, O_3	(✓) via $PM_{2.5}$, O_3
CO	✓				
Arsenic				✓	✓
Cadmium				✓	✓
Lead	✓		✓	✓	✓
Mercury			✓	✓	✓
Nickel				✓	✓
Benzene				✓	✓
Diesel exhaust				✓	✓
Formaldehyde				✓	✓
Dioxins and furans				✓	✓

All of the bodies considered impacts of secondary pollutants (both particles and ozone).

No documentation has been found that describes the derivation of the damage costs linked to ammonia or ozone for the UK.

4.5.2 Modelling of pollutant dispersion and chemistry and population exposure

There may be a desire to use local models of pollution dispersion to describe effects on exposure in the country where emissions occur. The situation where this is most likely to make a difference concerns the release of pollutants at low levels within cities (i.e. from road transport and domestic sources), which is clear from the results published for the UK by Defra, for which damage from the road transport sector varies by more than a factor 10 between rural releases and emissions in Inner London (Defra, 2015b). For application to industry, some form of regionalisation is likely to make very little difference to results for smaller countries, given that the damage from emissions from tall stacks needs to be aggregated over a very wide area. Indeed, the UK does not currently provide damage costs differentiated by region for industrial sources.

For larger countries there may well be merit in providing damage costs for different regions from industrial sources. In France, for example, a facility located in the Paris Basin is likely to have much higher damages than one in Brittany which is considerably more remote from the major European centres of population. Similarly in the UK, where a plant in the South-East of the country is much closer to the national major centre of population (London) and to major centres in continental Europe than one located in Scotland or Northern Ireland.

The UK has adopted an important constraint on analysis, limiting quantification of pollutant effects to the UK and ignoring those that occur in other countries. It is reasonable that a country would wish to assess the benefit of its actions on its own citizens, and other countries may wish to do the same. This can be easily facilitated through the use of the EMEP transfer matrices that were used to generate the EEA damage costs[42]. However, as the earlier discussion on the polluter pays principle and the preventive principle demonstrates, it is entirely unreasonable to fail to account damage in other countries when considering whether or not additional abatement is required at a site. The Frank Bold paper (Holland, 2017) reaches a similar conclusion, that pollutant damage, and hence the benefits of pollution control, should be accounted for wherever it occurs and this is logically certainly the case for EU level action.

Denmark, like the UK has used its own model (in this case the Danish Eulerian model) to assess pollutant dispersion. The model accounts for impacts within its own territory, but unlike the UK it does not restrict analysis to its own boundaries.

4.5.3 Quantification of impacts

The four distinct sources considered here (Denmark, UK, USEPA and EEA) use different sets of response functions. The functions adopted by Denmark, USEPA and the EEA are broadly similar in the range of effects considered, although the Danish set is based on the functions used in the CAFE (Clean Air For Europe) assessment rather than the later work for the Commission on the Thematic Strategy on Air Pollution which used the HRAPIE recommendations. The set of functions adopted for the UK largely follow recommendations of the Committee on the Medical Effects of Air Pollutants (COMEAP) which tends to be more conservative. Defra has in recent times, however, supplemented the COMEAP recommendations with further analysis of air pollution impacts on productivity drawing on a report by Ricardo (2014) and a report on chronic bronchitis (COMEAP, 2016).

It is notable that the dominant response function used in the analysis, for the quantification of long-term effects of PM exposure on mortality, is the same in all cases, a 6% change in mortality per 10 µg/m^3 $PM_{2.5}$. This estimate seems particularly robust, having changed little, if at all, since the earliest analyses of the impact back in the mid 1990s.

That said, there is some variation in other response functions used by each country for similar impacts. However, it seems that much of this is down to conclusions on the function list in each case being reached at different times (first USEPA, then WHO-Europe for EEA, then COMEAP for Defra) rather than a specific desire to take a national position. The pace of research is especially evident for NO_2. HRAPIE concluded that effects of chronic exposure to NO_2 should be quantified using a response function of 5.5% change in mortality per 10 µg/m^3, but with a cut-point for analysis of 20 µg/m^3 [43] and with a 30% reduction in impact to account for possible overlap between the functions for PM and NO_2. Subsequent review in the UK by COMEAP to be published late summer 2018 suggests a smaller response function and potentially greater overlap with PM, but no cut-point for the assessment. The consequence of this will be for the COMEAP recommendation to give higher damage estimates at low concentrations and lower damage at higher concentrations. A major reason for the differences is that there have been several significant studies published on NO_2 since 2013 when HRAPIE concluded (see e.g. COMEAP, 2018).

Whilst the EEA damage costs are based on the same set of response functions applied in all countries, they do account for variation in health status, using information on mortality rates, hospital admission rates, etc. from WHO.

It may be logical for US analysts to focus on US data, simply given the much larger literature available for the country compared to others. Following from this, it may be considered that response functions

[42] This has been performed by the present author in separate analysis for the Netherlands.

[43] In other words, quantification should not consider concentrations lower than 20 µg/m^3, a figure selected to recognise the lack of data below this concentration rather than as a perceived threshold for effect.

generated by a study in one's own country in Europe will be more accurate than those derived internationally, as they could implicitly take into account the effects of differences in underlying health status, for example linked to smoking behaviour and eating habits. However, experience shows that there can be important differences between studies carried out in the same country. As noted above, for the most important effect so far as economic assessment is concerned, the quantification of mortality linked to chronic exposure to fine particles there has been no significant change in the best estimate of response. This is despite the publication of numerous further studies, some of which have generated higher estimates, others lower. To move away from this best estimate (recognised as such by both USEPA and WHO-Europe amongst others) could well reduce the quality of analysis rather than improve it.

For non-health impacts, to crops, forests and materials, there is agreement on common sets of response functions for application across Europe. However, associated damage is small compared to health impacts.[44]

4.5.4 Valuation of quantified impacts

The largest contribution to damage costs comes from the valuation of mortality: this is common to the results from all of the organisations considered here. However, there are substantial differences in practice in the valuation of mortality between the EU, the UK and US EPA. Differences are also noted within different EU bodies. These positions (and others) are as follows:

- The EU uses a range for VOLY of €52,000 to €120,000 and a range for the value of statistical life (VSL) of €0.98 to €2.2 million (2005 prices). Values were derived from the EC-funded NewExt study, part of the ExternE series (IER, 2004). These values are used in sensitivity analyses, with no preference expressed for any individual value. The range as a whole leads to a factor 3 difference in damage cost estimates. The EEA report refers to both the low and high ends of the range (low VOLY to high VSL).
- A subsequent study, NEEDS, derived a lower estimate for the EU for the VOLY from surveys in a larger number of countries, of around €40,000 (believed to be 2010 price, but may be earlier) (Desaigues, 2011).
- For air pollution impacts the UK applies only the value of a life year (VOLY) adopting a value of around €33,000 in 2005 prices. The analysis that supported the adoption of this figure is currently under review. UK values are uplifted by 2% annually to account for economic growth and presumed increased earning.
- Denmark applies a higher value for the VOLY than either the EC or UK, of €76,000. Denmark also applies a higher value for the VSL, broadly similar to the upper bound used by the EC but lower than OECD.
- OECD carried out a meta analysis of mortality valuation studies, arriving at a recommended VSL for the EU of €3 million (2005 prices) (OECD, 2012). The advantage of this work compared to the NewExt study arises because of the large number of studies reviewed. On this basis, the conclusions should be considered more reliable. Although this work was funded by the European Commission, its recommendations have not yet been formally accepted for adoption in EC analysis (this is still under discussion). OECD does not recommend use of the VOLY as an alternative. The Nedellec and Rabl (2016) papers on toxic metal valuation adapted the earlier NEEDS estimate of the VOLY, increasing it by more than a factor 2, drawing on the conclusions of the OECD work, although Rabl was a lead author of the NEEDS analysis. Analysis under the REACH Regulation has tended to use values that are closer to the OECD VSL than the VSL used in the EC's analysis of air quality policy.
- Like OECD, US EPA only applies the VSL. Consideration was given to the use of the VOLY about a decade ago, but it was rejected on equity grounds, as it was perceived to bias against the elderly. The current recommendation is to apply a VSL of $7.4 million (2006 price,

[44] For crops and forests, see Holland, M. et al (2015a) and Holland, M. (2015b). For materials, see Bickel, P. and Friedrich, R. (2005).

equivalent to around €6 million) (US EPA, n.d.) , though there seems to be some variation and higher figures have been used.

The US figure is clearly higher than that used in Europe. This position can be supported in part by reference to higher per capita earnings in the US. It is also thought to be a function of the reliance of the US on the results of 'wage risk' studies, contrasted with the stated preference 'willingness to pay' approach preferred in Europe.

The question arises of whether one should use national value estimates for the analysis. For impacts where people in a given country would bear the costs of the policy in question, but also reap most of the benefits, use of national VSL values is correct. For impacts involving a significant transboundary element (e.g. CO_2 emissions and emissions of other air pollutants in all but the largest countries), it is better to use a common value.

There is general agreement that the discount rates used should be in the order of 4% (UK, for example, uses 3.5%). There is also some consensus that WTP estimates should be increased in future years to account for economic growth. The UK uses an annual factor of 2% for this, effectively offsetting about half of their discount rate.

4.6 Adapting damage costs to sectors

Whilst the focus of this work is the quantification of damage costs at the national level, there is a strong rationale for varying damage costs by sector when considering the benefits of the IED. National average damage costs that are undifferentiated by sector will introduce some bias to the analysis. Given that large industrial sites tend to be outside of the most densely populated urban areas, and release emissions from tall stacks, it is likely that they will generate lower damage per tonne emission than other sources such as transport that release pollutants close to ground in towns and cities. As noted already, the EEA provides information on sectoral adjustments in Annex 4 of their 2014 report. The UK accounts for differences in its damage costs for individual sectors, including waste, agriculture, the electricity supply industry, other industry, road transport and domestic, recognising that there will be variation in damage per tonne according to the type of emission source. This variation arises because of differences in population exposure to pollution from sources arising from:

- Variation in the distribution of population
- The height of emission release
- Pollutant plume characteristics

The methods proposed by the EEA report were based on a limited set of model runs. Ideally a more systematic approach would be taken, requiring a model such as EMEP to be run repeatedly to generate sector specific damage costs. For large countries it would also be beneficial to undertake a regionalised analysis.

4.7 Discussion

4.7.1 Health assessment: Best practice and recommendations for additional work

Table 4-2 provides recommendations for best practice and suggestions for additional work in quantification of the health impacts of the major local and regional air pollutants. Table 4-3 then provides a similar overview for the toxic metals and other trace pollutants. With respect to additional work it is recommended that priority is given to the items listed in Table 4-2 (especially those in bold) because these pollutants and effects are likely to dominate the analysis.

Table 4-2: Recommendations for assessment of health impacts associated with emissions of NH_3, NOx, $PM_{2.5}$, SO_2 and VOCs. Bold: actions identified as priorities for improving the quality of damage costs.

Step	Best practice	Additional work
Timescale	Quantify over the full timescale for which benefits are estimated to accrue	
Dispersion modelling	Account for continental scale, not only the country of origin. Account for pollutant chemistry. Apply factors to adjust national average damage costs to the sectors under investigation.	**Generate transfer matrices for individual sectors within each country, or review existing adjustment factors as used by the EEA.** **Regionalisation of transfer matrices for the largest countries (e.g. those lager than 150,000 km^2).**
Exposure assessment	Account for future population change with respect to size, age structure and health status	
Impact quantification	Adopt agreed position of WHO-Europe, as currently expressed via the HRAPIE recommendations. Use up to date information on incidence of disease in each country (from WHO).	**Supplement with additional information where compelling new evidence has become available. Ideally this would be led by WHO, drawing on a range of expertise similar to the work carried out in HRAPIE.**
Valuation	Apply the valuations adopted by Holland (2014) in assessment of the Thematic Strategy on Air Pollution. Ensure valuations are updated so that the price year for benefits matches that for costs (Roy and Braathen, 2017, section 4). Adopt a social discount rate of 4% (real price), reflecting EC practice, and allow for increased valuations in future years via economic growth.	**EC-led workshop on valuation estimates[45], considering specifically the VSL recommended by OECD from their 2012 meta-analysis with subsequent applications.**
Uncertainty assessment	Focus on the question of which uncertainties may be most likely to affect conclusions on the balance of costs and benefits	Generate a central catalogue of material on uncertainties in the assessment to provide guidance to stakeholders.

[45] The values currently used in EC air quality assessment are based on those agreed at a similar workshop held in 2001.

Table 4-3: Assessment of health impacts associated with emissions of trace pollutants including toxic metals. Bold: actions identified as priorities for improving the quality of damage costs.

Step	Best practice	Additional work
Timescale	Quantify over the full timescale for which benefits are estimated to accrue	
Dispersion modelling	Account for continental or global scale, not only the country of origin. Account for pollutant chemistry for organics and mercury. Account for cross-media transfers for pollutants for which ingestion is an important exposure pathway. Apply factors to adjust national average damage costs to the sectors under investigation, particularly for pollutants acting mainly via inhalation.	Review of dispersion modelling and information on cross-media transfers.
Exposure assessment	Account for future population change with respect to size, age structure and health status. Account for trends in diet affecting intake.	
Impact quantification	Apply the response functions adopted by EEA (2014), or by the European Chemicals Agency (ECHA) during assessment of proposed Restrictions or requests for Authorisation. Use up to date information on incidence of disease in each country (from WHO). Consider extent of possible overlap between quantification for the major local and regional air pollutants and for the trace pollutants.	**Keep response functions under review (this may be facilitated via ECHA).** **Agree whether economic assessments should use an extended list of health impacts in line with the papers by Nedellec and Rabl.**
Valuation	Apply the valuations adopted by EEA (2014). Ensure valuations are updated so that the price year for benefits matches that for costs. Adopt a social discount rate of 4% (real price), reflecting EC practice, and allow for increased valuations in future years via economic growth. Account for time lags between release and ingestion and inhalation.	EC-led workshop on valuation estimates[45], considering specifically the VSL recommended by OECD from their 2012 meta-analysis. Review of valuations adopted by and in work for, ECHA.
Uncertainty assessment	Focus on the question of which uncertainties may be most likely to affect conclusions on the balance of costs and benefits	Generate a central catalogue of material on uncertainties in the assessment to provide guidance to stakeholders.

Table 4-4: Quantification of impacts of NH_3, NOx, SO_2 and VOCs on ecosystems, including crops and forests. Bold: actions identified as priorities for improving the quality of damage costs.

Step	Best practice	Additional work
Timescale	Quantify over the full timescale for which benefits are estimated to accrue	
Dispersion modelling	Account for continental or global scale, not only the country of origin. Account for pollutant chemistry. Apply factors to adjust national average damage costs to the sectors under investigation, particularly for pollutants acting mainly via inhalation.	Review of dispersion modelling and information on cross-media transfers.
Exposure assessment	Account for all crops and forest production. Focus analysis of impacts on the exceedance of critical loads for nitrogen.	
Impact quantification	Follow recommendations of the Working Group on Effects under the Convention on Long Range Transboundary Air Pollution. Extend recommendations of the Working Group on Effects to all crops and all productive forest types, drawing on analysis carried out in the ECLAIRE study (Holland et al 2015a,b).	Keep response functions under review (this may be facilitated via Working Group on Effects, particularly the International Collaborative Programme on Vegetation). Keep information on exceedance of critical loads under review, drawing on periodic estimates issued by Working Group on Effects and by the GAINS modelling team at IIASA.
Valuation	Apply the valuations identified by Holland et al. (2015a,b) under the ECLAIRE study. Ensure valuations are updated so that the price year for benefits matches that for costs. Adopt a social discount rate of 4% (real price), reflecting EC practice, and allow for increased valuations in future years via economic growth.	Review new work on ecosystem valuation.
Uncertainty assessment	Focus on the question of which uncertainties may be most likely to affect conclusions on the balance of costs and benefits.	Generate a central catalogue of material on uncertainties in the assessment to provide guidance to stakeholders. **Specific consideration should be given to the reliability of valuations of ecosystems given the extent of exceedance of the critical load for nitrogen. If these valuations are considered unreliable, alternative means for factoring ecosystem damage into the analysis may be required in cases where health benefits alone do not yield a net benefit.**

The additional work recommended in these tables is not essential to the generation of country-specific damage costs for air pollutants: as noted elsewhere, these already exist and have not been outdated by subsequent work. However, the additional work that is recommended here would improve or maintain the quality of analysis in the years to come.

The four activities highlighted in bold in the tables, under the suggestions for additional work, have to a large extent previously been carried out either for the Commission or under the scientific support to the UN/ECE Convention on Long Range Transboundary Air Pollution, under general activities to improve European air quality rather than specifically in the context of industrial emissions. Discussion across different Units of the Commission should ensure that future work of this type is efficiently aligned with the needs of BAT assessment as well as other policy work. Specific suggestions of additional work that would be particularly beneficial to the work related to BAT would be:

- The extension of transfer matrices to include industrial sources separate from non-industrial, and regionalisation of matrices at a sub-country level for some of the larger countries as necessary.
- Updating of response functions for health for NO_2, ozone and $PM_{2.5}$, and extension of the function set to include metal and organic trace pollutants.
- Ensuring that there are not unjustified inconsistencies in valuation of endpoints in different policy areas.
- Agreement on approaches for integration of ecological damage alongside health impact assessment in the definition of VAT.

Precise estimates of the resources required for these activities are not possible here, though all of these activities would require significant input across a range of technical areas:

- Dispersion modelling to update and/or extend transfer matrices: would require involvement of experts in dispersion modelling with access to appropriate models. Development of the transfer matrices requires a large number of model runs on high speed computers. The transfer matrices currently used for policy related work for the European Commission were developed using the EMEP model.
- Update of HRAPIE recommendations on health response functions for air pollutants: HRAPIE, and the related REVIHAAP study, were major initiatives by the European Commission, involving a large number of experts (87) across the two studies:
  - Scientific advisory committee: 8 experts drawn from Europe and North American
  - Expert authors: 29 experts, again European and North American
  - External reviewers: 40 experts, European, Asian and North American
  - Observers: 4 staff from the Commission, integrated assessment modelling teams and industry
  - WHO secretariat: 6 staff

  The work involved review of an extensive literature and a series of meetings at WHO offices in addition to the writing of the reports. Total costs are estimated here to be between €250,000 and €500,000 (the actual budget for the earlier work should be available from WHO or the Commission). New work building on the conclusions of REVIHAAP and HRAPIE is unlikely to be any cheaper, especially if it is extended to additional pollutants.

- Update of health valuations: Could be completed through a 2 day workshop involving around 10 experts from Europe and North America covering the fields of both impact assessment (to provide information on what, precisely, needs to be valued) and economic valuation. One should allow a 6 day time allowance per expert for meeting preparation, participation and write up. Plus 10 days additional time required to manage the process and organise the workshop.

It may be considered appropriate to extend such a workshop to include other policy areas where valuation is used (e.g. chemicals policy, road safety, noise policy) to encourage consistency, and to understand any reasons that may exist for deviating from a common approach.

- Review of ecosystem valuations and debate of approaches for integration of ecological damage to the policy making framework: Again, a workshop format would be necessary to gain broad consensus on the way forward, whether it involves explicit valuation of ecological threats or alternative approaches. This would need to be informed by a study to define the broader policy framework around ecosystem protection, the policy tools that are currently used to assess threats and response, and the strengths and limitation of those tools.

Any more detailed estimation of budget would need to consider the extent to which activities could be factored into existing plans for research and analysis.

4.7.2 Generation of updated values and country-specific values

This section considers further work that may be undertaken to generate country-specific values specifically for use in BAT assessments. Additional work defined in the previous section would improve the quality of all the options detailed in this section, but is not essential to them, as demonstrated by the fact that country-specific values are available in the report issued by the EEA (2014). Although that work was completed 4 years ago the methods used largely reflect the current state of the art.

Three alternative options are identified for consideration by the European Commission to potentially support the development of national damage costs. All focus mainly on aspects related to health impacts as these provide the major part of damage estimates:

- Option 1: Centralised update of the existing modelling framework used to derive the EEA (2014) values
- Option 2: Develop modelling capabilities in each MS
- Option 3: Provide a simplified modelling system based on the existing framework that would enable individual MS to adjust specific parameters

Option 1: Update the existing modelling framework

A set of updated, nation-specific damage costs could be derived centrally following approaches of previous studies. This represents the least resource intensive option.

This option involves using the same or similar models used for derivation of the existing EEA estimates. These are:

- The ALPHA-RiskPoll model and extensions to it for ecosystem valuation made under the ECLAIRE study for quantification of impacts of emissions of NH_3, NOx, $PM_{2.5}$, SO_2 and VOCs)
- The RiskPoll model for quantification of impacts of toxic metals and organics
- Existing or updated transfer matrices derived using the EMEP model, or new transfer matrices derived using one of the alternatives listed in Section 4.3.2
- Outputs from the GAINS model, particularly for quantification of the areas at risk of critical loads exceedance.

All of these models are still in operation and most are being used to inform the process of policy development for the EC. They have all been subject to extensive review by stakeholders from Member States. Experts in three Member States (France, Sweden, UK) are now running national versions of ALPHA-RiskPoll. Alternatives include the Danish EVA (Economic Valuation of Air pollution) model, though it is not known how easy it would be to adapt that model to other parts of Europe.

The costs of this option are largely dependent on whether or not certain actions are covered under other existing or planned future EC activities.

For this option there are questions regarding ongoing maintenance of the modelling framework. The existing unit damage costs have been quantified under unrelated contracts. Given the demand for these figures and their policy relevance it would be beneficial to introduce a systematic maintenance programme for this work.

Table 4-5: Update of national damage costs under Option 1.

Step	Example models and required actions	Resources
Staff training in Member States	Production and use of training materials	With modelling to produce the unit damage costs being carried out centrally, staff training in each Member State would focus on understanding the modelling process and factors most relevant to national assessment. Staff would need to be familiar with the quantification process, but would not require in-depth knowledge.
Dispersion modelling	EMEP, CHIMERE, etc. Use of transfer matrices, but utilising ad-hoc factors drawing on past analysis to generate factors that illustrate variation of damage costs across different sectors and in different regions of larger countries.	If the existing transfer matrices and adjustment factors are to be used this phase requires no additional effort (assuming the existing models are retained). Additional effort would be required to integrate new transfer matrices.
Exposure assessment	Combine transfer matrices with population data and information on other sensitive receptors	This stage could be factored into the generation of transfer matrices.
Impact quantification and valuation	ALPHA-RiskPoll, etc. Apply models already developed for quantification of health and environmental benefits of changes in air pollution	Limited resource requirement, if ancillary activities including review of response functions continue to be carried out as part of other ongoing activities.
Uncertainty assessment	E.g. TUBA Framework Generation of central resources (tools and guidance) to enable consistent assessment and reporting of uncertainties and evaluation of their likely impact on the cost-benefit relationship.	Low resource requirement given availability of existing materials.
Model validation	Testing of model operation	Low resource requirement given availability of existing materials.

The expertise required for this option, and associated time requirements, are as follows:

- Tool development and maintenance: expert in air pollution modelling, with familiarity in emissions modelling, dispersion modelling, health and environmental impact assessment and economic valuation. Introduction of alternative assumptions as requested by Member States or the Commission. Time required: 30 days in the first year.
- Training of national experts by the model developer: 5 days in the first year for preparation of materials and holding of a training workshop.

- Further discussion and analysis by the model developer with national experts, responding to queries as required. 2-10 days, depending on how widely the tools are employed.
- National experts: understanding of air pollution impact assessment and economic valuation. For some countries this may be one person, for others, two. As a minimum, it is anticipated that they would require 5-10 days training per person in the first year, roughly 150 to 300 days effort across all Member States, though these costs would presumably be met by the Member States themselves (Additional costs associated with use of the damage costs are not considered here)
- Project management, including liaison with the Commission and national experts and reporting: 15 days in the first year.
- Total effort: 50-60 days in first year for centralised actions, plus 5-10 days training per Member State (150-300 days total across the EU), plus cost of a training workshop. Costs would be reduced in the second and subsequent years, roughly halved, but needing to be retained at a significant level to facilitate any necessary updates and to provide backup to experts in Member States. It is assumed that Member States would bear the costs incurred by their own experts.

Further inputs could be needed in subsequent years if new materials (e.g. revised response functions) are generated, and to respond to queries and suggestions made by the national experts. Decisions would need to be taken as to whether model development is carried out as a one-off activity, or whether maintenance should be scheduled in from the outset.

The main benefits and drawbacks of this option are as follows:

- Benefits
    - Lowest cost of the 3 options
    - Consistency across Member States
    - Transparency with respect to adjustments made to core set of values
- Disadvantages
    - Limited opportunity for Member States to change values
    - Some Member States may disagree with the models selected for key parts of the analysis

Option 2: Develop modelling capability in each country

A second option would be for each country that wished to generate its own estimates to develop their own modelling framework, drawing on guidance from existing tools. MS would follow pre-determined guidance and best practice steps, but would essentially replicate the activities carried out in all other MS in their own MS.

Each MS would require access to a dispersion model like EMEP or CHIMERE, or to transfer matrices generated by those models, and to maps showing the distribution of receptors across Europe. Analysis could use guidance provided alongside the 2014 cost-benefit analysis for the Thematic Strategy on Air Pollution for quantification of health impacts (Holland, 2014b) for application of the HRAPIE recommendations. The guidance provides information on the response functions that are recommended and background data on the incidence of ill health. Economic values were provided in the main CBA report (Holland, 2014a). A complete analysis would also require development of national tools for quantification of ecosystem and material damage, though a simple alternative may be possible from extrapolation of existing data[46].

[46] A simplified approach

This is the most expensive of the three options as it would require substantial duplication of effort across Europe, with staff in different countries developing similar modelling frameworks and being trained to understand the science underpinning the analysis to a high level. The technical expertise required for Option 2 is the same as Option 1 (e.g. expert in air pollution modelling, valuation of health impacts, etc) however this expertise must be available in each MS rather than in a centralised resource. Costs would depend on the expertise and existing tools available in any country.

This approach may also detract from transparency, with the reasons for differences in estimates generated by different countries being unknown.

Table 4-6: Update of national damage costs under Option 2.

Step	Required actions	Resources
Staff training	Production and use of training materials	The establishment of national expertise at a level that would enable the independent production of models would require a substantial amount of training.
Development of the modelling framework	Definition of data needs and sources Development of tools for processing of data.	Dependent on experience of the model developer
Dispersion modelling	Production of updated transfer matrices + Separation of industrial sectors and regions for large countries	If the existing transfer matrices and adjustment factors are to be used this phase requires no additional effort (assuming the existing models are retained). Additional effort would be required to integrate new transfer matrices. However, given the complexity of models that incorporate both atmospheric chemistry and long-range transport, it is considered likely that any country wishing to go down this route already has modellers in place.
Exposure assessment	Combine transfer matrices with population data and information on other sensitive receptors	This stage could be factored into the generation of transfer matrices.
Impact quantification and valuation	Identification of health risk functions and valuation data.	
Uncertainty assessment	Generation of tools and guidance to enable consistent assessment and reporting of uncertainties and evaluation of their likely impact on the cost-benefit relationship.	

The expertise required for this option, and associated time requirements, are as follows:

- Central activities could be limited to production and use of training materials, including guidance on modelling frameworks. The development of generic guidance and provision of support applicable to a range of models and disciplines will be more time consuming than providing guidance relating to a specific dataset or tool (estimate of 40 days in the first year, including time spent preparing for and presenting at a workshop).

- After the first year there may be no need for centralised input, with Member States working independently, though periodic updates to a core data set covering response functions, incidence of poor health and valuations would be useful. The potential for inconsistency in applications and assumptions may create a need for continual review of practice. Resources for such review could be significant if it is performed annually across all Member States. Allowing 2 days per Member State per year, total resource would be between 50 and 60 days per year.
- Costs to Member States would be significant, even for those that already have modelling frameworks in place. Several Member States already have dedicated teams running dispersion models already, and it would be sensible for additional work on impact assessment and valuation in those countries to be joined up with those existing teams. However, it would be necessary to add to those teams a high level of expertise in a range of disciplines: health and ecological impact assessment, environmental economics and uncertainty appraisal. As a minimum, it is anticipated that each Member State would need to commit 50 days to establishing the modelling framework in the first year, assuming that they have suitable dispersion models (including pollutant chemistry and operating at the full European scale) in place already and that those appointed to do the work are already familiar with much of the science. Across all Member States this would equate to 1400 days in the first year.
- Total effort: 40 days in first year for centralised actions, plus 50 days per Member State for model development (1400 days total across the EU), plus cost of a training workshop. It is assumed that Member States would bear the costs incurred by their own experts.

The main benefits and drawbacks of this option are as follows:

- Benefits
  - Development of modelling capacity in each country, enabling Member States to carry out analysis to their own sets of assumptions, increasing local acceptance of the analysis.
- Disadvantages
  - Most expensive of the 3 options when costs to Member States are considered
  - High potential for inconsistency across Member States
  - Reduced transparency and comparability between Member States
  - Unlikely to replace the need for a more centralised system, as some or many countries may not be in a position to develop their own tools and may require substantial support and guidance

Option 3: Provide a simplified modelling system based on the existing framework that would enable countries to adjust specific parameters

This option would adapt the current modelling framework to enable users in Member States to introduce alternative data, for example:

- Alternative pollution exposure estimates, either across the whole modelled region or for the emitting country
- Alternative data on the incidence of various health conditions
- Alternative risk factors for quantification of health risks
- Additional types of health impact and option to vary inclusion of non-health impacts
- Alternative valuation estimates.

This would be facilitated through a tool or set of guidance within which MS can flex parameters and input their own data to derive nation-specific damage costs. This represents a middle option in terms of resource requirements.

Table 4-7: Provision of a simplified modelling system under Option 3.

Step	Required actions	Resources
Staff training	Production and use of training materials and guidance	Could draw extensively on materials that have already been produced.
Development of the modelling framework	Discussion with stakeholders on the parameters that they would like to be able to adjust. Definition of data needs and sources. Development of tools for processing of data.	Limited cost for central model development assuming use of existing materials as a starting point. Costs for Member States and other stakeholders would be dependent on what variables they wish to provide alternative data for, and the extent to which they already have the necessary models or expertise in place.
Validation of the modelling framework	Testing to ensure the model operates correctly and Is straightforward for users.	In-house testing by 2 or 3 Member States varying in experience.

This represents a mid-cost option, between Option 1 (lowest cost) and Option 2 (highest cost).

The expertise required for this option, and associated time requirements, are as follows:

- Tool development and maintenance: expert in air pollution modelling, with familiarity in emissions modelling, dispersion modelling, health and environmental impact assessment and economic valuation. Much of this component of the work would be similar to that described under Option 1, but with the additional development of a user interface to permit Member States or the Commission to introduce alternative assumptions. Time required: 50 days in the first year including discussion with stakeholders.
- Training of national experts by the model developer: 5 days in the first year for preparation of materials and holding of a training workshop.
- Further discussion and analysis by the model developer with national experts, responding to queries as required. 2-10 days, depending on how widely the tools are employed.
- National experts: understanding of air pollution impact assessment and economic valuation. For some countries this may be one person, for others, two. As a minimum, it is anticipated that they would require 5-10 days training per person in the first year and an additional 2 days for participation at a preliminary workshop to assist in defining the user interface, roughly 210 to 350 days effort across all Member States, though these costs would presumably be met by the Member States themselves (Additional costs associated with use of the damage costs are not considered here)
- Project management, including liaison with the Commission and national experts and reporting: 15 days in the first year.
- Total effort: 70-80 days in first year for centralised actions, plus 5-10 days training per Member State (210-350 days total across the EU), plus cost of a training workshop and the workshop to define the user interface. Costs would be reduced in the second and subsequent years, roughly halved, but needing to be retained at a significant level to facilitate any necessary updates and to provide backup to experts in Member States. It is assumed that Member States would bear the costs incurred by their own experts.

As under other options, further inputs could be needed in subsequent years if new materials (e.g. revised response functions) are generated, and to respond to queries and suggestions made by the national experts. Decisions would need to be taken as to whether model development is carried out as a one-off activity, or whether maintenance should be scheduled in from the outset.

The main benefits and drawbacks of this option are as follows:

- Benefits
  - 2nd lowest cost of the 3 options
  - Provides a consistent framework across Member States
  - Provides opportunity for Member States to change values
  - Transparency with respect to adjustments made to core set of values provided that information on these adjustments is shared (this could be built into the interface)
- Disadvantages
  - Some Member States may prefer alternative models to be used for parts of the analysis.

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Appendices

Appendix 1: Results of prioritisation of all iron and steel process emission sources and pollutants for inclusion in scope of ex-post assessment

Appendix 2: High level estimates of emissions impacts – methodologies

Appendix 3: Integrated steelworks operator questionnaire

Appendix 4: Proforma for information collection from Member State competent authorities

Appendix 5: List of assumptions made per installation

Appendix 6: Coverage of the study

Appendix 7: Comparison of emission savings between high level estimates and CBA

Appendix 8: Completed assessment templates for studies assessed for CBA vulnerabilities

Appendix 1: Results of prioritisation of all iron and steel process emission sources and pollutants for inclusion in scope of ex-post assessment

Process	Significance	Emissions source and pollutants	Key emissions source (expert judgement)	BAT Monitoring	Air: Channelled/ diffuse emissions Water: Indirect/direct discharge (indirect = common WWTP)	Requirement (BAT-AEL means BAT and BAT-AEL)	Expected impact of BATC (expert judgement)	Comments	Agreed scope
SINTER PLANT	41 processes listed in VDEH	**Primary air emissions from sinter strands**							
		Dust	High	Continuous	Channelled	20. BAT-AEL	High	-	Include
		SO_2	High	Continuous	Channelled	22. BAT-AEL	High	-	Include
		NO_X	High	Continuous	Channelled	23. BAT-AEL	High	-	Include
		PCDD/F	High	Discontinuous	Channelled	24, 25. BAT-AEL	High	-	Include
		Hg	High (for some MS)	Discontinuous	Channelled	21. BAT-AEL	High	Hg content depends on iron ore quality, however same technology as for other pollutants	Include
		Secondary air emissions from sinter cooling and other relevant sources (discharge, crushing, screening, conveying)							
		Dust	Medium	Discontinuous	Channelled, but initial emissions might be diffuse	26. BAT-AEL	Medium - High	Example for impact: Installation of abatement system at an European installation with significant emission reduction.	Include
		Waste water							
		Suspended solids, COD, Heavy metals	Low	Qualified random sample or 24-hour composite sample	Direct	27, 28. BAT-AEL	Low	Expected that no longer applied in EU. Data availability concern - monitoring may only be undertaken at a point that does not coincide with BAT-AEL	Exclude
		BATs without BAT-AELs or BAT-AEPLs							
		Waste (production residues)	N/A	N/A	N/A	29, 30, 31. BAT		Agreed focus is on BAT-AELs	Exclude
		Energy	N/A	N/A	N/A	32. BAT		Agreed focus is on BAT-AELs	Exclude

Process	Significance	Emissions source and pollutants	Key emissions source (expert judgement)	BAT Monitoring	Air: Channelled/ diffuse emissions Water: Indirect/direct discharge (indirect= common WWTP)	Requirement (BAT-AEL means BAT and BAT-AEL)	Expected impact of BATC (expert judgement)	Comments	Agreed scope
PELLETISATION PLANT	7 processes listed in VDEH (6 SE, 1 NL)	**Air emissions from raw materials pre-treatment, induration strand, pellet handling and screening**							
		Dust	High	Discontinuous	Channelled	33. BAT-AEL	Unknown	All Member State 1plants are standalone installations (so E-PRTR might be more useful), attempt MS level assumptions via Member State 1EPA. NOx excluded as focus is on BAT-AELs.	Include
		SO_2	High	Discontinuous	Channelled	34. BAT-AEL	Unknown		Include
		NO_X	High	Discontinuous	Channelled	35, 36, 37 BAT (no AEL)	Unknown		Exclude
		HCl	High	Discontinuous	Channelled	34. BAT-AEL	Unknown		Include
		HF	High	Discontinuous	Channelled	34. BAT-AEL	Unknown		Include
		Waste water							
		Suspended solids, Kjeldahl nitrogen, Heavy metals, COD	Unknown	Qualified random sample or 24-hour composite sample	Direct	38, 39. BAT-AEL	Low	Only 7 plants) according to the BREF are already below BATC. Member State 2 data in BREF: in 2007 all parameter with the exception of suspended solids were below BAT-AELs. Suspended solids were <10-95mg/l against a BAT-AEL of <50mg/l.	Exclude
		BATs without BAT-AELs or BAT-AEPLs							
		Waste (production residues)	N/A	N/A	N/A	40. BAT	Unknown	Focus is on BAT-AELs	Exclude
		Energy	N/A	N/A	N/A	41. BAT	Unknown	Focus is on BAT-AELs	Exclude

Process	Significance	Emissions source and pollutants	Key emissions source (expert judgement)	BAT Monitoring	Air: Channelled/ diffuse emissions Water: Indirect/direct discharge (indirect= common WWTP)	Requirement (BAT-AEL means BAT and BAT-AEL)	Expected impact of BATC (expert judgement)	Comments	Agreed scope
COKE OVEN PLANTS	**56** processes listed in VDEH	**Coal grinding**							
		Dust	Low / Low - Medium	Discontinuous	Channelled	42. BAT-AEL	Low -Medium		Include
		Coal storage (if extraction and de-dusting is used)							
		Dust	Low -Medium	Discontinuous	Channelled if technique is applied	43. BAT-AEL	Low	BATC dependent on application of technique	Exclude
		Coal charging							
		Dust	Medium	Discontinuous	Diffuse	44. BAT-AEL	Low -Medium	Many of these diffuse sources are larger (coarse) particles --> lower health impact.	Include
		Coke quenching							
		Dust	Medium	Discontinuous	Most are Channelled	51. BAT-AEL	Low -Medium		Include
		Coke grading and handling							
		Dust	Low -medium	Discontinuous	Channelled and diffuse	52. BAT-AEL	Low -medium	Low impact and not key source	Exclude
		Coke pushing							
		Dust	Low -medium	Discontinuous	Most are Channelled	50. BAT-AEL	Low -medium	Low impact and not key source	Exclude
		Underfiring							
		NOx	High	Discontinuous	Channelled	49 BAT-AEL	Low -Medium		Include
		SO_2	High	Discontinuous	Channelled	49 BAT-AEL	Low -Medium		Include
		Desulphurisation of coke oven gas							
		Residual content of H_2S	High	Discontinuous	Channelled (no direct emissions)	48. BAT-AEPL	Medium	BAT-AEPL (not binding), not BAT-AEL. However, as has downstream impacts on processes which use COG, higher compliance likely.	Include
		Waste water							
		-Sulphides, Thiocyanate, Cyanide, PAH, Phenols, COD, BOD, ammonia-nitrogen	High	Qualified random sample or 24-hour composite sample	Indirect.	56. BAT-AELs	BAT referring only to single coke oven water treatment plants; if discharge is however indirect no need to include in data collection. Consider both standalone coke oven installations and integrated steelworks. In the BREF several reference coke oven installations are equipped with own waste water treatment plants.		Include
		BATs without BAT-AELs or BAT-AEPLs							
		Waste (production residues)	N/A	N/A	N/A	57. BAT		Focus is on BAT-AELs	Exclude
		Energy	N/A	N/A	N/A	58. BAT		Focus is on BAT-AELs	Exclude

Process	Significance	Emissions source and pollutants	Key emissions source (expert judgement)	BAT Monitoring	Air: Channelled/ diffuse emissions Water: Indirect/direct discharge (indirect= common WWTP)	Requirement (BAT-AEL means BAT and BAT-AEL)	Expected impact of BATC (expert judgement)	Comments	Agreed scope
BLAST FURNACE	**30** processes listed in VDEH	**Storage bunker of coal injection**							
		Dust	Low -medium	Discontinuous	Channelled	59. BAT-AEL	Low -medium		Include
		De-dusting of blast furnace gas - residual dust concentration							
		Dust	Medium - high	Discontinuous	Channelled (no direct emissions)	64. BAT-AEPL	Low	BREF states "*BF gas treatment is widely applied at blast furnaces around the world*." BAT-AEPLs are considered rather high.	Exclude
		Cast house							
		Dust	High	Continuous	Channelled, but initial emissions might be diffuse	61. BAT-AEL	Medium	Most installations expected to already have cast house dedusting (. Higher impact if not.	Include
		Diffuse emissions released during charging							
		Dust	Low - Medium	Discontinuous	Diffuse	63. BAT (no AEL)	Low - medium	See footnote[47]	Exclude
		Hot stoves							
		Dust	Medium	Discontinuous	Channelled	65. BAT-AEL	Low - Medium	Include these to get more comprehensive installation-wide SO2/NOx assessment, even though consider low /medium impact.	Include
		SO2	Medium	Discontinuous	Channelled	65. BAT-AEL	Low - Medium		Include
		NOX	Medium	Discontinuous	Channelled	65. BAT-AEL	Low - Medium		Include
		Waste water (Blast furnace gas treatment)							
		Suspended solids, Cyanide, Iron, Lead, Zinc	High	Qualified random sample or 24-hour composite sample	Direct	67. BAT-AEL	Medium - High	In the BATC there is no restriction for indirect discharge, therefore it seems that direct discharge is rather common, hence maybe more likely to have dedicated monitoring.	Include
		BATs without BAT-AELs or BAT-AEPLs							
		Waste (production residues)	N/A	N/A	N/A	68-69. BAT		Agreed focus is on BAT-AELs	Exclude
		Resource management	N/A	N/A	N/A	70. BAT		Agreed focus is on BAT-AELs	Exclude
		Energy	N/A	N/A	N/A	71-74. BAT		Agreed focus is on BAT-AELs	Exclude

[47] Information is available from VDEh on whether BF apply bell-less systems (53 out of 74 BF). This is the prerequisite for BAT63, however further information on the application of BAT (primary and secondary equalising II. gas or ventilation recovery system, III. use of blast furnace gas to pressurise the top bunkers) is not available from VDEh.

Process	Significance	Emissions source and pollutants	Key emissions source (expert judgement)	BAT Monitoring	Air: Channelled/ diffuse emissions Water: Indirect/direct discharge (indirect= common WWTP)	Requirement (BAT-AEL means BAT and BAT-AEL)	Expected impact of BATC (expert judgement)	Comments	Agreed scope
BOF PLANT	**16** processes listed in VDEH	**BOF gas recovery and cleaning - residual dust concentration**							
		Dust	Medium - High	Discontinuous	Channelled (no direct emissions)	75, 76. BAT-AEPL	Medium	Only BAT AEPL. Consider data availability may be low .	Include
		Secondary de-dusting of basic oxygen furnace, including hot metal treatment, BOF-related processes and secondary metallurgy (processes might be treated together or separately)							
		Dust	High	Continuous	Channelled	78. BAT-AEL	High		Include
		On-site slag processing							
		Dust	Medium	Discontinuous	Channelled	79. BAT-AEL	Medium		Include
		Waste water (Continuous casting and BOF gas cleaning (if wet processes are applied):							
		Suspended solids, Iron, Zinc, Nickel, Total chromium, Total hydrocarbons	Medium	Qualified random sample or 24-hour composite sample	Direct	80, 81. BAT-AEL	Low - Medium	Waste w ater comes not only from continuous casting, but also from w et de-dusting. According to the BREF many installations in EU apply w et de-dusting (4 reference plants are mentioned w ith dry de-dusting and 8 reference plants with wet de-dusting)	Include
		BATs without BAT-AELs or BAT-AEPLs							
		Waste (production residues)	N/A	N/A	N/A	82. BAT		Agreed focus is on BAT-AELs	Exclude
		Energy	N/A	N/A	N/A	83-86. BAT		Agreed focus is on BAT-AELs	Exclude

Process	Significance	Emissions source and pollutants	Key emissions source (expert judgement)	BAT Monitoring	Air: Channelled/ diffuse emissions Water: Indirect/direct discharge (indirect = common WWTP)	Requirement (BAT-AEL means BAT and BAT-AEL)	Expected impact of BATC (expert judgement)	Comments	Agreed scope
ELECTRIC ARC FURNACE	**185** processes listed in VDEH	**Primary and secondary dedusting**							
		Dust	High	Continuous/ Discontinuous	Channelled	88. BAT-AEL	Medium	Monitoring depending on size	Include
		PCDD/F	High	Continuous/ Discontinuous	Channelled	89. BAT-AEL	Medium	Monitoring depending on size	Include
		Hg	High	Continuous/ Discontinuous	Channelled	87. BAT, 88. BAT-AEL	Medium	Monitoring depending on size	Include
		On-site slag processing							
		Dust	Medium	Discontinuous	Channelled	90. BAT-AEL	Medium		Include
		Waste water (Blast furnace gas treatment)							
		Suspended solids, Iron, Zinc, Nickel, Total chromium, Total hydrocarbons	Medium	Qualified random sample or 24-hour composite sample	Direct	91-92. BAT-AEL	Low - Medium	As important as BOF plant waste water treatment, so should be consistent with that.	Include
		BATs without BAT-AELs or BAT-AEPLs							
		Waste (production residues)	N/A	N/A	N/A	93. BAT		Agreed focus is on BAT-AELs	Exclude
		Energy	N/A	N/A	N/A	94. BAT		Agreed focus is on BAT-AELs	Exclude
		Noise	N/A	N/A	N/A	95. BAT		Agreed focus is on BAT-AELs	Exclude

Appendix 2: High level estimates of emissions impacts – methodologies

Method 1: Reported emissions 2012-16 compared to 2016 projection based on 2012 emissions and adjusted for activity changes

Description

A simple assessment of the change in emissions is simply the emissions reported for the identified installations in a year prior to the impacts of BATC implementation (say, 2011 or 2012[48]) compared to a year of reported emissions data for the latest year of data available, post implementation of the BATC requirements. This simple comparison is possible, but because it cannot assess how much of the change between the pre- and post-implementation year is attributable to changes in activity levels (production) of the sector, it limits the conclusions that can be drawn.

To account for this limitation, Method 1 compares the emissions reported in 2016 to an estimate of emissions in 2016 that are generated from applying the trend in activity data between 2012 and 2016 to the reported 2012 emissions. Activity data used were from the PRODCOM database in Eurostat, giving EU28 annual steel production; these are shown in Table A-1 and show small variations between years (Eurostat, 2017).

Table A-1: Eurostat steel production data from PRODCOM database (Eurostat, 2017)

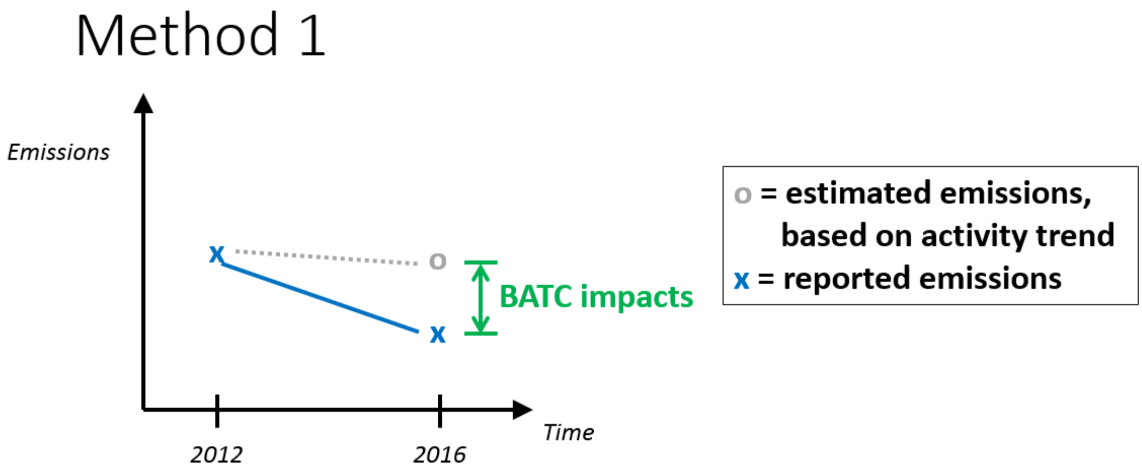
Production	2011	2012	2013	2014	2015	2016
Steel production (kt)	136,891	142,917	140,304	139,050	138,334	133,450

The emissions data used for Method 1 were extracted from E-PRTR (EEA, 2017b) for activities 1.3, 2.1 and 2.2, and from CLRTAP (EEA, 2017a) for sector code 2C1. Production quantities in 2012 were 142,917 kt, while production quantities in 2016 were 133,450 kt. Therefore, to project emissions from 2012 to 2016, 2012 emissions were multiplied by the ratio 133,450/142,917 = 0.934.

The emission reductions attributed to application of BAT are taken to then be the difference between the estimated 2016 emissions and the reported 2016 emissions. This is illustrated in Figure A-19.

[48] Although the BATC were adopted in 2012, and thus required to be implemented four years later in 2016, the requirements were well-known a while in advance of 2012, i.e. when the BREF was in the final draft stage. It is conceivable that operators may have acted prior to 2012 to comply with the anticipated BAT-AELs.

Figure A-19: Schematic of Method 1



Analysis was conducted comparing reported emissions trends from 2012 to 2016 with these projected emissions. This should give some indication of the extent to which application of BAT has affected emissions. The projected emissions give an indication of what emissions would have been in the absence of BATC, as they are projected based only on change in activity from the situation in 2012. Reported emissions by contrast should reflect all changes that have occurred in this period, including activity changes and changes as a result of the implementation of BATC.

Results

Tabular results are shown in Table A-2 and Table A-3.

The results of Method 1 are shown using emissions data from CLRTAP in Figure A-20. In the case of CLRTAP data, PCDD/F emissions have dropped lower than reported emission trends from 2013 onwards, although reported emissions rose from 2013 to 2015, they remain below projected emissions for the entire period. At least some of this change is likely to be attributable to implementation of BAT. This is not reflected in emissions of NOx, with reported emissions rising above the projected emissions. In the case of dust emissions, reported emissions rose above projected emissions from 2013-2014, falling in 2015 to similar levels as projected emissions, and finally falling below projections in 2016. This may reflect the influence of the BATC halting the rise in emissions and leading to a period of declining emissions. SO_2 and Hg emissions are similar in the with and without BATC scenarios. As previously noted, emissions reported under LRTAP are not subject to the same reporting thresholds as E-PRTR and as such reported emissions in each dataset are not directly comparable. Results of this method using E-PRTR data suggest more of an impact of BATC than when using LRTAP data, and this may be due to this difference in thresholds, with E-PRTR corresponding more closely to the scope of IED installations.

Other limitations of E-PRTR data have been summarised in Ricardo (2018) as:

- The completeness of reporting across countries varies, and also across media. This is particularly the case for emissions to water: the EEA assessed completeness of 2014 data, concluding that across the EU a total of 3,627 facilities reported emissions to water out of the total 33,084 facilities reporting to E-PRTR.
- Many facilities report inconsistently over time, i.e. no report one year, and a positive report in a subsequent year.

- There are many water pollutants, such as persistent organic pollutants (POPs) which are very rarely reported by installations in E-PRTR.
- Facility operators and Member States may implement the E-PRTR reporting rules inconsistently. This may be an issue in particular for the correct allocation of water emissions depending on whether these are discharged or transferred to an external waste water treatment plant (WWTP). Further work on this topic is being carried out at the time of writing (March 2018) by Ricardo for the European Environment Agency in an ongoing study entitled "Off-site transfers and releases of wastewater: state of play and their treatment in Europe".
- Where a facility carries out multiple activities that fall within the remit of the E-PRTR Regulation, it is only required to report its total emissions against its main activity (as defined in the Regulation). Where multiple activities are carried out in one facility, it may be difficult to distinguish the source of emissions and thus there may be inaccuracies in the reporting arising from source attribution.

The results of applying Method 1 using air emissions data from E-PRTR are shown in Figure A-21. In the case of E-PRTR data, implementation of BATC would appear to have had a more marked effect, with reported emissions dropping below projected emissions in the case of SOx, NOx and Hg, with particularly large drops in 2016 in the case of PM_{10}, and from 2015-2016 in the case of NOx, SOx and mercury. Whilst the number of facilities reporting to E-PRTR each year has reduced, it is unclear whether this is due to facilities having reduced emissions to below the reporting threshold or if the facility is closed and thus not reporting.

The fact that E-PRTR shows more marked reductions in emissions compared with projections than LRTAP may be due to the dataset being aligned more closely with IED installations and being based on actual reporting and so better reflecting the influence of BATC. LRTAP data reported by countries can include estimated or actual data – this situation varies by country and by sector. If the LRTAP figures are estimates rather than data, one can still expect these to reflect the BATC changes.

Figure A-20: Comparison of LRTAP reported emissions to air 2012-2016 with emissions projected from 2012 using Eurostat production quantities (Method 1)

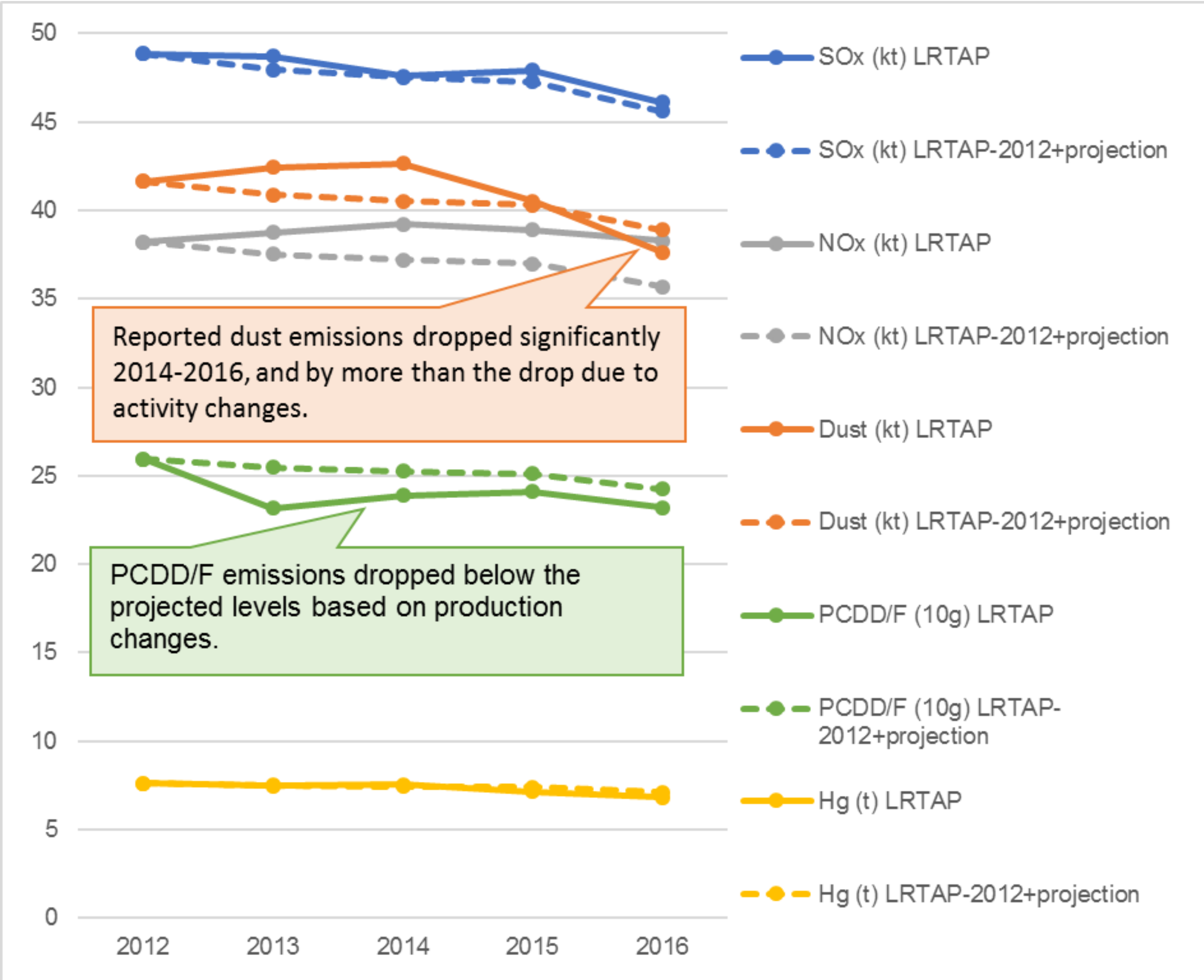
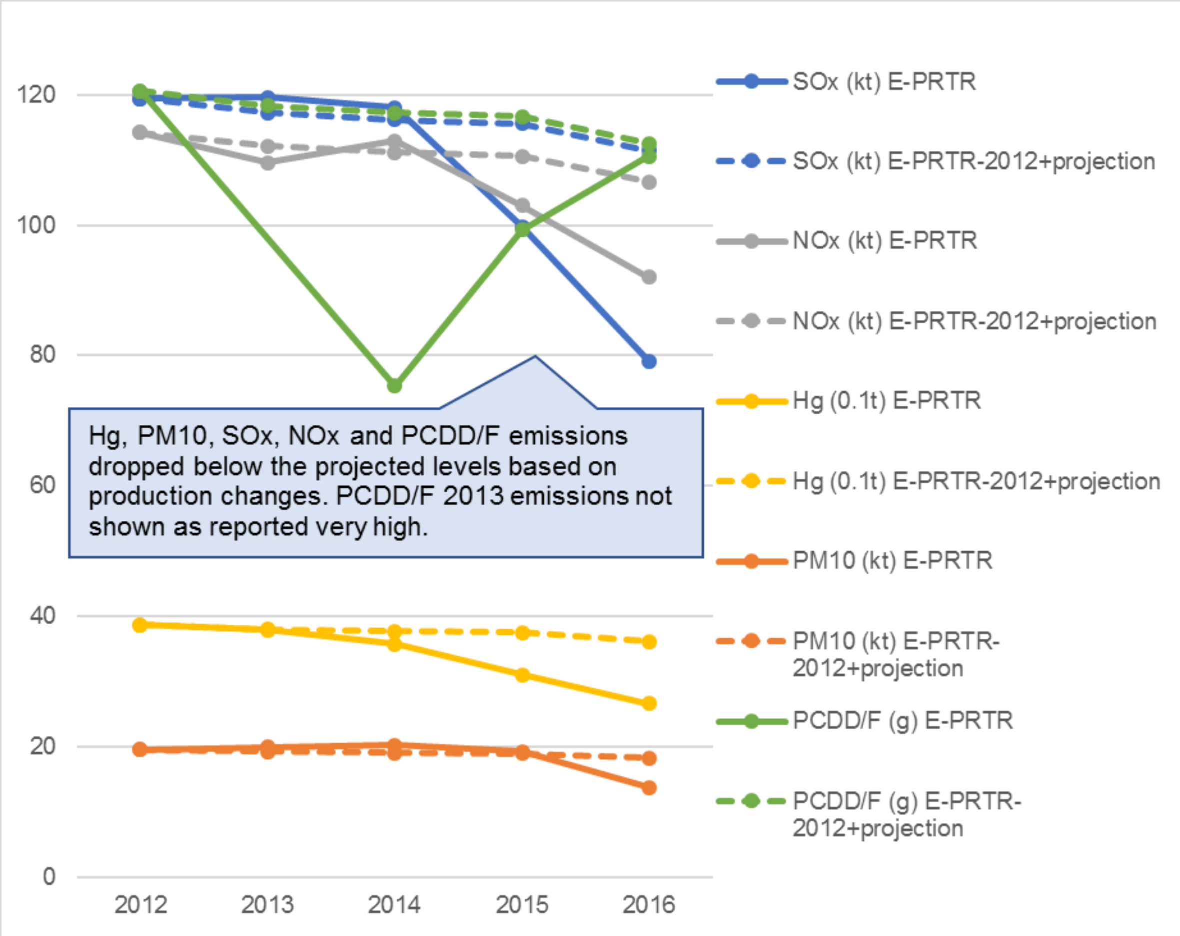


Figure A-21: Comparison of E-PRTR reported emissions to air 2012-2016 with emissions projected from 2012 using Eurostat production quantities (Method 1)



Similar analysis was conducted for emissions to water in E-PRTR in for pollutants identified as high priority for the sector. While reported emissions did drop below projected emissions for many pollutants, there was not sufficient facilities reporting emissions to water to have confidence in the result. In addition to this, the number of reporting facilities decreased over the studied period. Facilities reporting emissions of mercury to water was just 6 in 2012 and 6 in 2016, and facilities reporting emissions of arsenic decreased from 21 in 2012 to 14 in 2016. As such, emissions to water analysis is not shown graphically; Table A-3 includes the water results for 2015.

Table A-2: Emissions to air impacts of the BATC according to Method 1 (source: this study)

Pollutant	Unit	CLRTAP				E-PRTR			
		Without BATC (2016)	With BATC (2016)	Impact of BATC (2016)	% impact of BATC	Without BATC (2016)	With BATC (2016)	Impact of BATC (2016)	% impact of BATC
SO_2	(kt)	45.6	46.1	0.5	+1%	111.5	79.1	-32.4	-29%
NOx	(kt)	35.7	38.3	2.6	+7%	106.6	92	-14.6	-14%
Dust (LRTAP)/ PM_{10} (EPRTR)	(kt)	38.9	37.6	-1.3	-3%	18.3	13.7	-4.6	-25%
Hg	(t)	7.1	6.8	-0.3	-4%	3.61	2.66	-0.95	-26%
PCDD/F	(g)	242	232	-10	-4%	112.6	110.6	-2	-2%

Table A-3: Emissions to water impacts of the BATC according to Method 1 (source: this study)

Pollutant	Units	Without BATC (2015)	With BATC (2015)	Impact of BATC (2015)	% impact of BATC
Arsenic	kg	559	450	-109	-20%
Cadmium	kg	1,600	305	-1,295	-81%
Chlorides (total Cl)	kt	102	95	-6.8	-7%
Chromium	t	14	4.8	-10	-67%
Copper	t	9.5	3.3	-6.2	-65%
Cyanides (total CN)	t	52	18	-34	-66%
Fluorides (total F)	t	691	283	-407	-59%
Halogenated organic compounds (as AOX)	t	17	10	-7.2	-42%
Lead	t	14	5.2	-8.7	-63%
Mercury	kg	32	14	-19	-58%
Nickel	t	22	5.2	-17	-76%
Phenols (as total C)	t	22	27	5.4	25%
Polycyclic aromatic hydrocarbons (PAHs)	kg	1,376	315	-1,061	-77%
Total nitrogen	kt	4.2	3.3	-0.9	-21%
Total organic carbon (TOC) (as total C)	kt	2.3	2.6	0.3	11%
Total phosphorus	t	16	22	5.7	35%
Zinc	t	125	110	-15	-12%

Method 2: Results from Amec Foster Wheeler (2015) study

Description

Method 2 uses high-level estimates of emissions reductions from implementation of BATC from the iron and steel sector analysis carried out in the Amec Foster Wheeler (2015) study for the European Commission entitled "*Assessment of the potential emissions reductions delivered by BATC adopted under the IED*". Supporting this published report was an unpublished spreadsheet based analysis for several sectors, including iron and steel. In this unpublished spreadsheet, the aggregated emission data for the iron and steel sector were split by Amec using assumptions to process level to match the BATC granularity. This study compared a Business-as-Usual scenario with an IED scenario and base year and target years (including 2020), comparing emissions and specific emissions (e.g. tonnes of emissions per unit of activity or fuel use). This spreadsheet with underlying data has been provided by the Commission and has been used directly without updating for newer E-PRTR emissions as the analysis already accounted for emissions prior to BATC implementation.

Results

Results were outputted as absolute changes (e.g. t/year) and specific emission savings (mass of pollutant per tonne of product) in the year 2020. These results are aggregated and not at process level. Absolute emission reductions are shown in Table A-4, and specific emission reductions in Table A-5. There are large percentage reductions in absolute emissions for many pollutants due to the implementation of BAT, with the largest reductions in emissions to air of PCDD/F (86%), NO_x (53%) and SO_x (29%), and emissions to water of Pb (38%). In some cases, there are smaller changes in emissions, such as a 0.7% reduction in Hg emissions to air, and a 1% reduction in emissions of CN to water. In the case of HF and HCl, emissions are predicted to increase by 11% and 2%.

Table A-4: Estimated absolute emissions impacts of BATC in 2020 (Method 2) (Source: Amec foster wheeler, 2015)

Medium	Pollutant	Units	Without BATC (2020)	With BATC (2020)	Impact of BATC (2020)	% impact of BATC
Air	Dust	t	20,764	17,129	-3,635	-18%
Air	NOx	t	102,624	48,115	-54,509	-53%
Air	SOx	t	114,056	81,004	-33,053	-29%
Air	Hg	kg	4,057	4,030	-27	-0.7%
Air	PCDD/F	g	227	32.1	-194.6	-86%
Air	HF	t	212	234	+22.3	+11%
Air	HCl	t	915	929	+14.1	+1.5%
Water	Total N	t	561	488	-72.7	-13%
Water	CN	t	117	116	-1.2	-1.0%
Water	Pb	t	8.6	5.3	-3.2	-38%
Water	Zn	t	157	126	-30.3	-19%
Water	Ni	t	22.7	18.8	-3.9	-17%
Water	Cr	t	14.5	12.2	-2.2	-15%
Water	Σ As, Cd, Cr, Cu, Hg, Ni, Pb, Zn	t	30.8	28.8	-2.0	-6.6%

Table A-5: Estimated specific emissions (per tonne of production) impacts of BATC in 2020 (Method 2) (Source: Amec foster wheeler, 2015)

Medium	Pollutant	Units (/t product)	Without BATC (2020)	With BATC (2020)	Impact of BATC (2020)	% impact of BATC
Air	Dust	t/t	30.5	25.17	-5.34	-18%
Air	NOx	t/t	151	70.7	-80.1	-53%
Air	SOx	t/t	168	119	-48.6	-29%
Air	Hg	kg/t	6.0	5.9	-0.04	-0.7%
Air	PCDD/F	g/t	0.33	0.05	-0.29	-86%
Air	HF	t/t	0.31	0.34	+0.03	+11%
Air	HCl	t/t	1.35	1.37	+0.02	+1.5%
Water	Total N	t/t	0.83	0.72	-0.11	-13%
Water	CN	kg/t	172	170	-1.8	-1.0%
Water	Pb	kg/t	12.6	7.9	-4.8	-38%
Water	Zn	kg/t	230	186	-44	-19%
Water	Ni	kg/t	33.4	27.6	-5.8	-17%
Water	Cr	kg/t	21.3	18.0	-3.3	-15%
Water	Σ As, Cd, Cr, Cu, Hg, Ni, Pb, Zn	kg/t	45.3	42.3	-3.0	-6.6%

Method 3: Emission reduction factors using BAT-AELs

Description

Emissions data were extracted from E-PRTR for IED activities 1.3, 2.1 and 2.2 for the year 2012. These were combined to provide total emissions for the sector. The total emissions for the sector from E-PRTR were projected to 2016 using the trends in production data. Production data was derived from the publication European Steel in Figures (EUROFER, 2017) and the 2016 Steel Statistical Yearbook (Worldsteel, 2016), as well as the PRODCOM database in Eurostat (Table A-6). Two production datasets were used to check for sensitivity.

The potential impact of the BATC on emissions was estimated by applying emission reduction factors based on the difference between IPPC BREF upper BAT-AELs and the IED BREF upper BAT-AELs. In the case of several scoped in pollutants/processes, there were no IPPC BAT-AELs and as such reduction estimates were not possible for these pollutants: for example, there is no IPCC BREF SOx BAT-AEL for blast furnaces. As IPPC BAT-AELs were not required to be implemented in installation permits, the reductions from this method may be underestimates.

The emission reduction factors were applied to total facility emissions; in instances where multiple processes were scoped as high priority for a given pollutant, reduction factors of both processes were applied as upper and lower estimates. This is a rather coarse approach as the E-PRTR-reported emissions will include all on-site emission sources (e.g. including LCPs). It likely excludes EAFs as these installations may not be captured by E-PRTR due to the reporting thresholds.

This method is summarised in Figure A-22.

Figure A-22: Schematic of Method 3

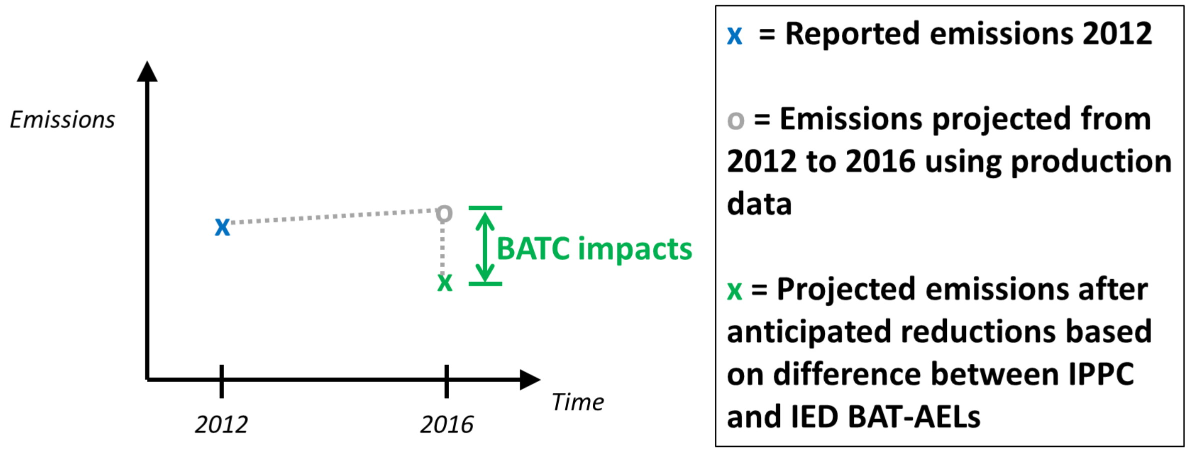


Table A-6: Crude steel production quantities used in Method 3

Year	Crude steel output from integrated steelworks (kt)	Total crude steel output (kt)	Source
2010	-	172,911	Worldsteel (2016)
2012	142,917	168,756	Eurostat (2017), EUROFER (2017)
2013	140,304	-	Eurostat (2017)
2014	139,050	-	Eurostat (2017)
2015	138,334	166,191	Eurostat (2017), EUROFER (2017)
2016	133,450	161,979	Eurostat (2017), EUROFER (2017)

Emissions reduction factors from the comparison of IPPC and IED BAT-AELs for pollutants identified in the scoping stage are shown in Table A-7. There are high percentage reductions from IED AELs compared to IPPC AELs in the case of NOx emissions from blast furnaces (71%), dust emissions from sinter plants (70%) and PCDD/F emissions from sinter plants (60%). In limited instances, upper BAT-AELs in the IED BREF are higher than the IPPC BREF, such as dust in blast furnaces (33%) and nitrogen emissions to water in coke ovens (66%); in these instances, no change is assumed (i.e. effectively assuming the IPPC AEL was not achieved in practice). In the case of mercury and NMVOCs, no reduction estimates were made due to lack of AEL for any process in the IPPC BREF.

Table A-7: Comparison of upper BAT-AELs and resulting emission reduction factors

Process	Source	Prioritised Pollutant	Medium	IPPC upper-AEL	IED upper-AEL	Units	Emissions Change
Sinter plant	Primary	Dust	Air	50	15	mg Nm^{-3}	-70%
	Primary	PCDD/F	Air	0.5	0.2	ng I-teq Nm^{-3}	-60%
	Primary	SO_2	Air	500	500	mg Nm^{-3}	-
Coke oven	Underfiring	NO_X	Air	770	500	mg Nm^{-3}	-35%
	Waste water	Cyanide	Water	0.1	0.1	mg/l	-
	Waste water	Ammonia-nitrogen	Water	30	50	mg/l	Zero assumed
Blast furnace	Hot blast stove	Dust	Air	10	10	mg Nm^{-3}	0
	Casting house	Dust	Air	15	20	mg Nm^{-3}	Zero assumed
	Hot blast stove	NOx	Air	350	100	mg Nm^{-3}	-71%
BOF	Secondary dedusting	Dust	Air	15	15	mg Nm^{-3}	-

Results

The results of applying the emission reduction factors to 2016 projected emissions are shown in Table A-8, Table A-9 and Table A-10. The range in estimates are produced by using emission reduction factors of different processes.

Table A-8: Method 3 estimated NOx emissions.

Nitrogen oxides (NOx/NO2)	Without BATC (2016) (kt)	With BATC (2016) (kt) (Note 1)
Projecting with WorldSteel/ EUROFER production data	120	34 – 78
Projecting with Eurostat production data	115	33 – 75

Note 1: The range is from multiple emission reduction factors. Lower: blast furnace AEL reduction factor. Upper: coke oven AEL reduction factor

Table A-9: Method 3 estimated dust emissions.

Dust	Without BATC (2016) (t)	With BATC (2016) (t) (Note 1)
Projecting with WorldSteel/ EUROFER production data	20,289	6,087 – 20,289
Projecting with Eurostat production data	19,414	5,824 – 19,414

Note 1: The range is from multiple emission reduction factors. Lower: sinter plant BAT-AEL reduction factor. Upper:: blast furnace hot stove BAT-AEL reduction factor.

Table A-10: Method 3 estimated PCDD/F emissions.

PCDD/F (as Teq.)	Without BATC (2016) (g)	With BATC (2016) (g)
Projecting with WorldSteel/EUROFER production data	121	24.2
Projecting with Eurostat production data	116	46.3

Appendix 3: Integrated steelworks operator questionnaire

Operator
Questionnaire 2901

Appendix 4: Proforma for information collection from Member State competent authorities

Steelworks BATC impacts - MS author

Appendix 5: List of assumptions made per installation

Confidential information

Appendix 6: Coverage of the study

Process type	Number of processes in EU	Number of processes for which information on BATC impacts known	Proportion of sector covered by the study		Number of impacted processes	Proportion of processes impacted, in sample	Proportion of processes impacted, in EU
			By number	By capacity			
Sinter strand	38	23	61%	68%	13	57%	34%
Pelletisation	7	6	86%	84%	1	17%	14%
Coke oven	53	26	49%	55%	13	50%	25%
Blast furnace	71	44	61%	57%	8	19%	11%
BOF	32	19	59%	62%	3	16%	9%
EAF	197	125	63%	65%	40	32%	20%

Appendix 7: Comparison of emission savings between high level estimates and CBA

Pollutant	Method	Year	Sensitivity	Units	Impact of BATC
SO2	Method 1	2016	CLRTAP	kt	+0.5
SO2	Method 1	2016	E-PRTR	kt	-32.4
SOx	Method 2	2020	-	kt	-33
SO2	CBA Model	2016	-	kt	-13.9
NOx	Method 1	2016	CLRTAP	kt	+2.6
NOx	Method 1	2016	E-PRTR	Kt	-14.6
NOx	Method 2	2020	-	kt	-55
NOx	Method 3	2016	Eurostat, blast furnace AELs	kt	-82
NOx	Method 3	2016	Eurostat, coke oven AELs	kt	-40
NOx	CBA Model	2016	-	kt	-0.5
Dust	Method 1	2016	CLRTAP	kt	-1.3
PM10	Method 1	2016	E-PRTR	kt	-4.6
Dust	Method 2	2020	-	kt	-3.6
Dust	Method 3	2016	Eurostat, sinter plant AELs	kt	-13.6
Dust	Method 3	2016	Eurostat, BF casting house AELs	kt	0
Dust	CBA Model	2016	-	kt	-8.0
PM10	CBA Model	2016	-	kt	-4.7
Hg	Method 1	2016	CLRTAP	t	-0.3
Hg	Method 1	2015	E-PRTR	t	-1.0
Hg	Method 2	2020	-	t	-0.03
Hg	CBA Model	2016	-	t	-0.5
PCDD/F	Method 1	2016	CLRTAP	g	-10
PCDD/F	Method 1	2016	E-PRTR	g	-2.0
PCDD/F	Method 2	2020	-	g	-195
PCDD/F	Method 3	2016	Eurostat	g	-69.7
PCDD/F	CBA Model	2016	-	g	-12.9

Appendix 8: Completed assessment templates for studies assessed for CBA vulnerabilities

Study
review_consolidatec

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