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# How many stones to kill a bird?

## A Case Study of Policy Interactions within the Netherlands NOx Emission Trading Scheme.

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# Abstract

The purpose of this study is to investigate the impact of the co-existence of overarching ‘command and control’ regulatory policies on the cost-effectiveness of subordinate Emission Trading Systems (ETS). The Netherlands introduced an ETS in 2005 as a cost-effective measure to decrease the emission of Nitrogen Oxides (NOx) in stationary sources. The ETS was repealed after eight years, however, because of the limited effectiveness of the scheme. The influence of the Integrated Pollution Prevention and Control Directive (IPPC), a regulatory standard adopted to achieve industrial emission objectives at the European level, was anticipated to be a cause of this limited effectiveness. The Dutch NOx ETS was, therefore, chosen as the focus for this study as it allows key questions to be addressed about the influence of vertical interactions between overarching climate regulations and Emission Trading Schemes on the cost-effectiveness of the trading scheme.

The influence on the cost-effectiveness of the ETS was assessed by calculating the specific costs of the actual implemented abatement techniques in terms of EUR/kg NOx abated and comparing these to the remaining least-cost abatement potential. Data from the emission reports of all participants of the NOx ETS, as provided by the NEa, was used to assess the abatement techniques. The abatement techniques not found in the emission reports were found by means of a company survey and cost functions in the literature were used to assess the specific cost of each measure.

The results, explaining 17.5 ktons of the total 20.2 ktons NOx NOx abatement achieved in 2012, showed that 16 ktons abated NOx emissions were attained at relatively low-cost. Since virtually all abatement originated from companies subject to the IPPC, it is clear that the directive undermined the functioning of the emission market. Because of the interaction between the Directive and the NOx ETS, 1 kton of abatement was attained at significantly higher cost than remaining least-cost abatement potential and thus caused estimated additional abatement costs of 10 million EUR per year.

Compared to the regulatory cost of the ETS, however, the additional abatement costs were shown to be relatively limited. The IPPC Directive succeeded in enforcing a large proportion of abatement at relatively low cost, thereby reducing the gain in cost-effectiveness achievable by an ETS with no interactions. The reason for this was found to be due to the sensitivity of the IPPC emission limits to the cost-effectiveness of abatement measures and the relative homogeneity of the cost of NOx abatement techniques.

# Contents

# Acronyms and definition

| | |
|---|---|
| NOx | Nitrogen oxides |
| ETS | Emission trading scheme |
| SCR | Selective catalytic reduction |
| SNCR | Selective non-catalytic reduction |
| LNB | Low NOx burners |
| ULNB | Ultra low NOx burners |
| DLE | Dry Low NOx emission system |
| DLN | Dry Low NOx emission system |
| Installation | Individual unit within a site which can either by monitored individually or in clusters (*installatie*): |
| Site | An activity on a commercial scale which is active within certain boundaries and is included in annex 1 of the *Besuit Omgevingsrecht* as stated in the Wet Milieubeheer (Dutch Environmental Protection Act |
| CAC | Command and Control policies |
| IPPC | Integrated Pollution Prevention and Control, later integrated in the IED |
| IED | Indutrial Emissions Directive |
| GJ | Gigajoules |
| MW | Megawatt |
| $MW_{th}$ | Megawatt thermal capacity |
| $MW_{e}$ | Megawatt electric capacity |
| PSR | Performance standard rate in grams NOx/GJ or tons product |

# 1
# Introduction

Air pollution emissions have been a constant concern ever since the industrial revolution. Anthropogenic emissions contribute to global warming, threaten natural ecosystems and are detrimental to human health. Although in Europe there has been a significant reduction in the level of many air pollutants over the last decades, concentrations are often still too high, while in upcoming economies such as China, air pollution concentrations remain critical and pose an ever increasing threat to human health (European Environment Agency, 2016). The only way to ensure the reduction of these toxic emissions is through effective policy. Finding both the most efficient and cost-effective emission abatement strategy is, therefore, essential

One significant group of damaging air pollutants are nitrogen oxides. Nitrogen oxides (which includes NO and $NO_2$, and collectively referred to as $NO_x$) are acidifying compounds formed by the combustion of fuels at high temperatures. NOx is damaging to the health of both humans and ecosystems as the compound readily reacts with other compounds to form nitric acid, ozone and other toxic products. However, the primary sources of nitrogen, i.e. the oxidator that forms NOx, is the nitrogen content of the air and fuel used in combustion processes. As a consequence, the source of nitrogen is difficult to regulate. Nonetheless, a wide range of NOx control technologies are available and the best available technologies (BAT) can reduce emissions to very low levels. The focus of both national and international regulations regarding air pollution emissions have been on increasing the development and deployment of such BAT measures.

In 2005, the Dutch government implemented an emissions trading scheme (ETS) as a NOx emission abatement strategy. The scheme ran until 2013 but was repealed when the administrative burden was found to outweigh the minimal effect of the policy (DHV, Van der Kolk Advies, & Hofland Milieu Consultant, 2007). The emission trading scheme covered all emission sources from combustion processes of over 20 $MW_{th}$ and included various industrial sectors with process related emissions. In contrast to the *cap and trade* design of the EU Carbon Emission Trading Scheme, the NOx ETS was designed as a relative emission trading scheme in which emission trading rights were earned through a relative benchmark, known as the performance standard rate (PSR). This benchmark allocated emission rights based on either a quantity of NOx per gigajoule (GJ) of fuel input or tons produced product. Total emission trading rights of companies were calculated by multiplying the PSR by the actual fuel input or production in a certain trading year. The rationale behind the ETS was that the emitting facilities that could reduce emissions cheaply, would become net sellers of emission trading rights to firms with higher marginal abatement costs thereby minimizing the costs over the whole scheme since firms would implement the cheapest abatement options would be.

Whilst the policy itself was not seen as a success, emission trading schemes in general are regarded as effective instruments for meeting the demand for policies with a high degree of cost-effectiveness. From a welfare economics perspective, the emissions problem is a classic case of market externalities. There is a clear disparity between the social costs of emissions endured by society and the private costs of the emissions producing agent. The agent does not consider the social cost of emissions in its cost function and, as a consequence, produces more goods and, therefore, more emissions than is socially optimal. Because of ready availability of abatement techniques, national emissions policies have a tendency to focus strongly on the forced implementation of emissions abatement technologies through so called *command and control (CAC)* regulations. However, economic theory dictates that forcing *all* actors to reduce their emissions ignores differentiation in abatement costs and will not lead to the minimization of social costs (Perman, Ma, McGilvray, & Common, 1996). Designing the most efficient policy to reduce air pollution emissions at the least cost is seen as a challenge in the policymaking arena, a view reinforced in a large body of scientific literature.

The NOx emission trading scheme, which theoretically should have achieved cost-effectiveness but didn't, serves to demonstrate that although the economic rationale behind an emission trading scheme may be simple, policy design is crucial for both cost- and environmental-effectiveness (Burtraw, Palmer, Bharvirkar, & Paul, 2001). An additional complicating factor is that since emission trading schemes are relatively new policy instruments, they are often implemented in an already existing policy landscape. The interaction that can occur among the plurality of environmental policies can lead to a different effectiveness than initially intended (Taschini, 2009). In the case of the Netherlands, their $NO_x$ trading scheme was subordinate to ruling European emission limits set by the Integrated Pollution Prevention and Control Directive (IPPC), later integrated in the Industrial Emissions Directive (IED). Although the permit market was effectively formed sufficient supply and demand participants participated in the trading scheme, the market was confronted with an oversupply of permits and little price forming occurred. Although the trading scheme has been evaluated in several studies coordinated by DHV, the exact effects of overreaching EU policy has not been quantified. Kroon (2003), showed that the European IPPC has little influence on the implementation rate of abatement techniques for non-ETS participants but suggests that this influence on ETS participants may be stronger.

The IPPC regulations required that all environmental permits issued to companies covered by the directive, set emission limits at the upper end of through BAT attainable emission levels. It is hypothesised that the integration of these emission limit values reduces the flexibility of companies to implement the least-cost abatement options. If the IPPC emission limits forced these companies to implement abatement techniques they would not otherwise have done under a sole ETS scenario, it would have led to a loss of efficiency and cost-effectiveness of the national policy.

## 1.1 Scope and research question

Because the Dutch NOx Emission Trading Scheme was the first emission trading scheme to be implemented within the EU to tackle conventional pollutants, the European Commission (EC) has granted permission for it to be used as a case study (Van der Kolk Advies, Emission Care, & DHV, 2011). The nesting of European policies was seen to be one of the factors reducing the effectiveness of the policy and further investigation into the effect of the policy interactions (PI) of trading schemes with an overlapping policy is warranted. Since the Dutch NOx emission trading scheme was subordinate to other emission trading schemes, it provides an interesting case to gain a better understanding of the influence of the interaction in a hierarchy of policies. The following specific research questions will be studied:

*What is the effect of policy interactions of the European IPPC directive on the cost-effectiveness of the Dutch NOx emission trading scheme (ETS) and how does this impact future policy designs with similar ETS designs?*

This study is structured as follows: section two provides the theoretical basis of emission trading schemes within the context of policy interaction. Section three describes the methods used to evaluate the interactions of the IPPC regulation and the NOx ETS. Section four presents the results of the study into the cost-effectiveness of the NOx emission scheme and provides an evaluation of the abatement potential in a scenario without IPPC regulations.

Section five presents the discussion and evaluation of the results in order to arrive at an assessment of the influence of policy interactions in a general context. Included in section five are the limitations and assumptions of the research. The study concludes with a conclusion and recommendations for further study.

# 2

# Literature review

The two main concepts central to the research are cost-effectiveness of emission trading schemes and policy interactions (PI). This section discusses these concepts and provides a basis for the methods described in the section following. First the welfare economic perspective of cost-effectiveness is dealt with followed by a discussion on policy interaction.

## 2.1 Cost-effectiveness of ETS

Emissions in welfare economic theory hold both damages and benefits (Perman et al., 1996). From an environmental perspective, the damages of emissions are more evident than their benefits. Whereas damages can be calculated as being the damage of a pollutant to either anthropogenic or natural systems, the benefits can be viewed as the ability of firms to continue to produce goods that cause emissions. Not being allowed to emit any pollutant would lead to high abatement costs or the inability to produce the goods at all. Firms thus benefit from not being required to reduce their emissions to zero.

Because of the existence of both damages and benefits, an optimal level of emission can be found by maximizing the gap between total benefits and total damages. The optimal level of the emission of a pollutants is the point where the net social benefits (NB) are maximized; the net benefits are equal to the benefits (B) minus the damages (D) of pollutant Z, as seen in function one.

$$NB(Z) = B(Z) - D(Z) \tag{1}$$

The optimum can be found by setting the marginal net benefits, the first order derivative of NB(Z), to zero. The optimum can then be found where the marginal benefits of emission are equal to the marginal damages, as seen in function two and three.

$$\frac{\mathrm{dNB(Z)}}{\mathrm{dZ}} = \frac{\mathrm{dB(Z)}}{\mathrm{dZ}} - \frac{\mathrm{dD(Z)}}{\mathrm{dZ}} = 0 \tag{2}$$

$$\frac{\mathrm{dB(Z)}}{\mathrm{dZ}} = \frac{\mathrm{dD(Z)}}{\mathrm{dZ}} \tag{3}$$

As the marginal benefits of pollution are similar to the marginal abatement costs, the same first order condition holds for the optimal abatement level. Minimized total cost of pollution will be where the marginal abatement costs are equal to the marginal damages of pollution. Increasing abatement further will outweigh the abatement cost to pollution damage cost and vice versa and thus no Pareto improvement can be made from that point (Perman et al., 1996). This first order condition defines the efficient distribution and policies achieving this condition are economically efficient.

To reach the optimal pollution level, policy is needed to incentivize firms to reduce their emissions. To reach an economically efficient abatement level, the cheapest abatement options should be chosen first (Perman et al., 1996). This condition states that at the efficient abatement level, the marginal abatement costs of all firms are equal. If they would be unequal, firm A with higher marginal abatement costs could reduce their abatement (increase emissions) and firm B with lower marginal costs increase their abatement. This would lead to a Pareto improvement if firm A were to compensate firm B for the endured costs (Tietenberg, 2003).

Currently, the following three policy instruments are widely used to increase the abatement of firms: command and control (CAC), emission taxation and tradeable permit-type policies (Perman et al., 1996). In CAC policies, non-transferable emission limits are set for each emitting facility. Because all firms are given an emission limit, the focus is not on attaining the least costs and the use of these policies can lead to cost-inefficiencies. Since all polluters are required to abate, the abatement is indifferent to the marginal abatement costs incurred by each individual firm and no equalization of the marginal abatement costs takes place. On the other hand, CAC policies do score high on environmental effectiveness because they allow the regulatory authorities to control emissions efficiently across regions. Kaswan (2011), however, warns that the cost-inefficiency of CAC policies should not be overstated. Within these policies, industrial standards with high differentiation between various installations can also differentiate between cost-effectiveness of measures between installations and are therefore able to enforce low-cost measures.

Market based policies leaves the market to differentiate the marginal costs between firms. Seen from this perspective, taxation and market-based permits can be more cost-efficient as market trading can ensure the adoption of the least cost measures. In the case of taxation, a tax can be set to reduce the marginal benefits of the pollution and internalize the costs of the pollution in the cost function of firms. If the value of the tax matches the marginal damage at optimal pollution level, it will ensure the emissions produced meet the optimal pollution level (Perman et al., 1996). Since firms choose their abatement level according to their marginal abatement costs (Elkins & Baker, 2002), either they will choose to abate to the point where their marginal abatement cost is equal to the tax or they will choose not to abate if their marginal costs are higher than the tax. By setting an efficient tax level an effective internalization of the pollution costs can be achieved.

Unlike taxation which sets a price and allows firms to decide on the level of abatement, a market-based permit system sets the optimal quantity of pollution and leaves the market to set the price. In this system, permits are issued which allow firms to emit one unit of pollutant. By setting a cap on the amount of permits available, firms will buy permits up until the permit price is equal to their marginal abatement costs, which will equal the market clearing price (Sijm & Van Dril, 2003). It is not relevant here how the permits are allocated: they can be either allocated freely (grandfathered) or auctioned off. If all permits are auctioned, firms will place their bid at the price of their marginal abatement costs and the total demand curve will equal the marginal abatement cost curve (Tietenberg, 2003). Because there is a cap, an equilibrium price will be found and firms will again equate their marginal abatement costs. If all permits are allocated based on past emissions (grandfathered) and traded on an open market, firms with higher marginal abatement costs will buy permits and the total equilibrium price will be equal.

While how the permits are allocated would be theoretically indifferent to their final distribution, it is argued that auctioning leads to the greatest efficiency gains(Elkins & Baker, 2002). Revenues from the auction can be used to reduce distortionary pre-existing taxes and will not leave profits of the permits to private firms and lead to windfall profits. The reduction of both pre-existing taxes and emissions is referred to as the ‘double dividend hypothesis’ (Boemare & Quirion, 2002). Furthermore, if the allocation of permits is based on grandfathering, firms are less incentivised to search for eco-innovations to reduce emissions as the decreased demand for permits such innovations would bring about would decrease the value of the firm’s commodity (Boemare & Quirion, 2002). The efficiency reducing effects of free allocation has been shown in several modelling exercises (Burtraw, Palmer, Bharvirkar, & Paul, 2001; Carmona, Fehr, Hinz, & Porchet, 2010).

In addition to the choice of allocation mechanism, three other factors determine the type of emission trading scheme. These three factors are the type of tradable commodity traded, the scope of the market and regulation, and the type of emission target (Sijm & Van Dril, 2003). Together these factors determine the class of ETS (Sijm, 2004). With the emission rights being the foundation of an ETS, the ETS can either be designed with allowances or with credits as tradable commodities. In credit systems or baseline and adjust systems, a baseline is set for the emitting activities of the emissions of participants. Emissions reductions causing the emission of an actor to be below their baseline are awarded credits, which can then be sold to firms whose emissions exceed their baselines (Perman et al., 1996). These baseline and adjust systems differ from cap and trade systems which allocate firms with emission allowances based on the allocation mechanism. These two systems essentially differ in what the commodities represent: credits represent a quantity of emissions reduced whilst allowances represent a quantity of emissions. The NO ETS is an example of a scheme with emission allowances, not credits.

The emission allowances or credits can either be awarded to upstream or downstream actors, thus determining the scope of the market and regulations (Sijm, 2004). This distinguishes what group of actors, either being the producers or consumers, are obliged to surrender the commodities to the competent authorities For instance in an ETS for carbon emissions, either the producers of fossil fuel or the consumers of the fuels can be the designed participants. For downstream systems, a further distinction is made in direct or indirect schemes, differing in the way the offsite generated electricity and heat is accounted for. In the NOx ETS, the producers of electricity are the participants of the scheme, consumers are not awarded or asked to surrender emission rights.

Lastly, the type of emission target can be either a relative target or an absolute target. An absolute target will set an emission limit in tons of emission output, whilst a relative target will limit the emissions either per the energy content of fuel input or tons product output (Sijm, 2004). The PSR of the NOx ETS clearly shows that the scheme has a relative target.

While the above mainly discusses the short-term theoretical notion of the cost-efficiency of trading schemes, these policies also have a number of more dynamic and long-term effects on innovation and technological diffusion. Scotchmer (2011) notes that while all barriers against emissions will increase technological diffusion, not all environmental policies will have like effects. For instance, under an ETS policy, the most innovative technologies and those most likely to lead to huge reductions in emissions could be hampered since they would also bring about a significant reduction in the permit price and thus in their value for private firms. This effect could be exacerbated in cases where several actors have market power.

The uncertainty in future price under an ETS has a technology reducing effect too, which should be given the necessary attention in the policy design (Taylor, 2012). Tax schemes, on the other hand, can have a positive effect on technological diffusion since the reduction of emissions will not hinder the commodity value of companies. CAC policies can cause a significant increase in the adoption low emission technologies, for instance, by setting mandatory BAT limits, and thus overcoming other market failures. The incentive could however cause firms to adopt BATs without looking any further and as result, such policies could lead to fewer long-term benefits. That being said, the effects of the initial adoption of BATs could reverberate to bring about further learning and other indirect effects (Perman et al., 1996).

Several of the large scale emission trading schemes that have been implemented (e.g. the EU Carbon ETS and the US Clean Air Act $SO_2$ scheme), were designed to surpass regulatory or national boundaries. In such cases, the relevant national or regional abatement policies are subordinate to the emission trading scheme. Intraregional trading is, however, not a necessary condition for an emission trading scheme, as demonstrated by the Dutch ETS.

An underlying assumption in the cost-saving potential of emission trading schemes is cost heterogeneity. If marginal abatement costs across emitters is equal and no economies of scale are present, an emission scheme cannot achieve greater costs savings than CAC policies. The environmental outcome in this case would be equal for both CAC and trading policies and, therefore, economists could still argue the case for market-based policies. Greater cost heterogeneity can increase the cost-savings of an emission trading scheme, relative to a CAC policy (Newell & Stavins, 2003). However, given the high regulative pressure on trading policies and the higher cost of transactions, this is not necessarily the case. Newel & Stavins (2003) describe this relationship in a model and show empirically that as a result of cost heterogeneity in abatement techniques, the US NOx Emission Trading Scheme achieved over 40% cost-reduction compared to CAC policies. Carlson, Burtraw, Cropper, & Palmer (2000) also explain the reduced gains from trading within the US $SO_2$ emission trading scheme as being due to increased cost homogeneity. Conversely, Ben-David et al. (1999) found a negative relation between cost heterogeneity and cost efficiency. Their research noted that increasing heterogeneity led to greater price variability in the market which in turn gave rise to suboptimal decision-making and mis-allocation. It is uncertain, however, whether the conclusions of Ben-David et al. (1999) can be extrapolated into general theory or if it is context specific to the experimental setting.

The theory on the efficiency of the scheme is not a good fit for pollutants which have regional effects and as a result are not uniformly mixed (Farber, 2012). In the case of uniformly mixed pollutants, it does not matter where the pollutants are emitted, the marginal damages will be equal to all. With non-mixed pollutants, the marginal damages are a function of the distance to the pollution source. As emission trading focusses on the implementation of abatement technologies where marginal abatement costs are low, it is conceivable that pollution will concentrate in areas where the marginal costs are higher and create pollutant hot-spots. This raises the issue of social justice as a trade-off is made between economic efficiency reaching the lowest level of emissions at the lowest cost of all and higher pollutant levels for some (Kaswan, 2011). Moreover, as formula 1 showed that cost efficiency is reached when the marginal damages equals the marginal abatement benefits, economic efficiency is not reached with undifferentiating caps as the marginal damages benefits is not uniform over all regions. This issue is discussed in both environmental justice as well as economic literature (Antweiler, 2015; Farber, 2012; Kaswan, 2011). Some of economic instruments used to tackle this problem are including zonal systems where inter-zonal trade is faced with different exchange rates and adopting heterogeneous prices for individual polluters within the same ETS (Antweiler, 2015). Kaswan (2011), however, opts for options such as CAC regulations within emission trading scheme to ensure an increase in environmental justice.

The potential efficiency increase of ETS is, among others, dependent on the functioning of the market and the existence of potential market failures. It is thus of no surprise that additional governmental policies can affect cost-effectiveness. The influence of these policies on the cost efficiency of ETS is an example of policy interactions and is discussed in the next section.

## 2.2 Policy interactions

From the economic theory described, additional policies within an ETS can lead to sub-optimal abatement and decreased cost-effectiveness due to policy interactions. Additional policies will encourage emitters to base their choice of abatement technologies not only on their own marginal abatement costs and thus opt for more expensive options. For instance, if within an ETS, subsidies would be provided for renewable energy (RE) generation, the carbon price would be lowered and non-renewables would be favoured (Philibert, 2011).

The interaction between additional policies and emission trading schemes, can, however, both enhance and reduce the environmental and cost effectiveness of the policy (Sorrell & Sijm, 2003). When additional policies address market failures within the emission market, a positive effect could occur, whilst overlapping policies might give stakeholders the double taxing effect or lead to cost inefficiencies (Sorrell & Sijm, 2003). These policy interactions are further complicated due to emission trading schemes being part of a *hierarchy* of policies where other policies can either overlap or be subordinate to the ETS.

As example of hierarchic policy interactions in emission trading schemes, Sorrell & Sijm (2003) discuss the $CO_2$ abatement target setting within the United Kingdom, whilst participating in the EU Carbon Emission Trading Scheme (EU ETS). The national target and subsequent policies as the abatement subsidies, will lead to decreased private abatement costs in the UK and thus increase the national abated emissions. The total EU cap, however, will remain equal and total EU abatement will not increase. Moreover, the total cost-effectiveness might decrease since more expensive options in the UK would be subsidized at the cost of cheaper options elsewhere. This example shows that the emission trading scheme is the top tier policy which is effected by underlying, national policies. Within the literature on policy interactions within emission trading schemes, a lot of focus is placed on this hierarchical structure while little attention is paid to the effects of the trading schemes as nested, lower tier policy, with intra-national regulations as the top tier policy.

The above UK case serves as an example of PI with a fixed cap and trade scheme which has a detrimental effect on the efficiency of the ETS. Proponents of multiple policies would state that an emission trading policy exclusively would be efficient in a perfect competitive market without market failures, and evidence suggests that a failure free market is not the case (Philibert, 2011; Sijm & Van Dril, 2003). In PI literature, a view held by various researchers is that "effective environmental policy requires policy instrument mixes rather than single policy instruments" - killing one bird with two stones in other words (Weber, Driessen, & Runhaar, 2013, p. 1381). Multiple environmental policies are justified in order to correct market failures other than the pollution externality or to serve other policy purposes. For instance, a subsidy on RE generation could enhance positive learning externalities, decrease cost of generation and further enhance the adoption within the ETS cap, decreasing societal costs.

As ETS' are relative newcomers and are often implemented as additions to, not replacements of the current environmental policies, the environmental policy landscape has become 'congested' (Oikonomou, Flamos, Zeugolis, & Grafakos, 2012). This congestion leads to environmental and non-environmental having mutual effects on each other, which are referred to as policy interactions (Oikonomou et al., 2012). Policy interactions are thus the effects that policies have on the effectiveness and efficiency of other policies. From the PI literature, it is shown that policy interactions can be either "complementary, competitive or self-exclusive"(Oikonomou & Jepma, 2008). Depending on the aim and goal of the policy and the way the policies interact, the interaction can lead to an enforcing or deteriorating effect on the cost-effectiveness of the policy.

Analysing the PI of trading schemes and multiple policies is especially relevant as the additional policies can have strong effects on the cost-effectiveness. Sorrell & Sijm (2003) show, for instance, in a partial equilibrium framework that UK emission obligations within the ETS will lead to decreased cost-effectiveness due to these PI. Within this context, stronger member state emission limits within an ETS are not justified as the total emission cap is unaffected. Additional policies, such as specific technology subsidies, aimed at reducing market failures or correct for negative externalities, could however be justified.

To study these interactions, Oikonomou & Jepma (2008) have constructed an analytical framework for these PI. In their framework, they identify interactions that can be either *inherently complementary*, *inherently counter-productive* or context-specific and specify different types of interactions. These interaction types can refer to the interactions of solely national policy, defined as *horizontal* interactions, or to *vertical* interactions which are interactions of international policies. They also distinguish *internal* and *external* interactions, which they define as the interactions between climate policies or climate and non-climate policies, respectively. Other interaction types are *operational, sequencing* and *trading* where operational interactions refer to policies that have a simultaneous active influence on actors, sequencing interactions are policies which follow each other in time and trading interactions are policies that influence each other through the trading of exchangeable commodities. The types of interactions of tradable permits are further broken down to the degree of exchangeability, which can be either one way, doubly interchangeable or fully separate. Finally, the policy interactions can be identified according to the level of specified interaction, where different policy measures can be either fully integrated, integrated for specific interactions or fully separate.

After identifying the types of interaction, the framework identifies the areas of interactions (Oikonomou & Jepma, 2008). From the works in the EU INTEACT project (Sijm and van Dril (2003) and Sorrell and Sijm (2003)), Oikonomou & Jepma deduced several areas where interaction can occur within the policy, these being at the: measure identification, objectives, scope, market arrangements, market flexibility, financing, technological parameters, timing compliance parameters and institutional arrangements. The aim of identifying these areas is to breakdown the different instruments and see where the instruments overlap, complement each other or are neutral (Oikonomou & Jepma, 2008).

As a third step, the framework evaluates the policy interactions. This can be done along the lines of effectiveness, efficiency, impacts on energy and market prices, impacts on society and innovation. As the framework is intended for ex-ante analysis, several methods are proposed for constructing a baseline to assess the interactions. The methods are trend extrapolation, econometrics, linear programming and judgement. The researchers, however, do state that these methods can be used for an ex-post analysis, as the methods can then be used to construct *'hypothetical scenarios without policy interactions'.* This is ex-post analysis, providing an evaluation of interactions within the NOx emission trading scheme, is the scope of this study.

While the framework of Oikonomou and Jepma prescribes several quantitative approaches for assessing policy interactions, most studies focussing on PI in practice take a more qualitative approach. Moreover, few works have been published on PI in which actual empirical data was used rather than theorizing on the effects of the interaction of several policies (Weber et al., 2013). For instance, in their research Sorrell & Sijm (2003) and Philibert (2011) focussed on a more theoretical level where the interactions are analysed in an economic partial equilibrium framework. Oikonomou et al., (2012) and Río (2010) also focus on more qualitative assessments of PI in energy policies and support schemes, without using empirical data. Kautto, Arasto, Sijm, & Peck (2012), too, take a generally qualitative approach although they also use empirical data gathered from literature reviews and expert interviews to assess whether the expected (theoretical) effects were actually observed. Weber, Driessen, & Runhaar (2013) also mainly focussed on empirical data, basing their ex-post evaluation on data gained from years of experience of noise pollution policies although, again, they also used literature for theoretical expected effects and expert interviews to analyse the *perceived* effects of the policy. Unlike this study which focusses on single policy instruments, Weber et al., (2013) analyse the whole policy package including *all* policy measures.

## 2.3 Hypothesis of the effects of the IPPC on the NOx emission scheme

In contrast to similar CAC policies, the IPPC-directive provides relative flexibility in that it does not dictate the implementation of BAT technologies, but rather places limits on the emission levels produced by these technologies. This leaves firms free to exploit alternative technologies as long as they are able to achieve at least equivalent or lower emission levels. The emission level values are laid down in the BAT conclusions in the BAT reference documents (BREF) and set the conditions the competent authorities should take up in the environmental permits awarded to firms. A BREF document is available for various industries in the EU and BAT conclusions are differentiated according to installation categories within each industry. The IPPC directive is, however, only relevant for installations that fall under the directive. The industries categorized as IPPC installations are identified in Annex 1 of the IED; the main relevant industries for the NOx emission trading are as follows:

- Combustion of fuels in installations rated higher than 50 $MW_{th}$;
- Refining of mineral oil and gas;
- Production of cokes;
- Production and processing of metals;
- Mineral industry;
- Manufacture of glass;
- Chemical industry;
- Waste management.

Within the trading scheme, most of these industries are also identified as process industries in which emission rights of the process oriented installations are calculated through a process PSR. Emission rights for installations solely producing heat or electricity and not being linked to a production process were calculated through the combustion PSR.

The installations within the NOx trading scheme which were not subject to stringent IPPC control are those between 20 and 50 $MW_{th}$ and the metal and mineral industries with capacities that fall below IPPC limits. Other industries, including the chemical industry, also are excluded from the IPPCD if the production volume is below a certain level, but as in general, the production volumes of these industries is very high, it is expected that all these companies meet the minimum levels (Uylenburg et al., 2012).

As previously described, the theory states that in an emission permit system with a limited amount of permits and no emission limit values, the firms with the lowest marginal abatement costs will abate. The addition of an extra policy could then only lead to an increase in cost-effectiveness if the policy were also to address failures within the market and correct for externalities. The difference between emission limit values and emission trading schemes, is that emission limits impose an obligation to abate whereas trading schemes offer an economic incentive to abate. While this rationale would be of no consequence in a market with purely rational actors, it is not unreasonable to speculate that this difference could be of consequence. Empirical evidence shows that firms do not always succeed in reaching all ‘low-hanging fruit’, even when environmental abatement techniques could have a positive net present value (Isaksson, 2005). An obligatory policy could force companies to reach for these fruits whereas an emission trading scheme may not provide sufficient incentives. Other market failures are however not evidently addressed by the IPPCD.

By obliging all firms to ensure their emissions were monitored accurately, the Dutch NOx emission trading scheme did succeed in addressing market failures that were due to a lack of information This measure had the added result that firms gained increased insight into the state of their emissions relative to the performance standard rate(DHV et al., 2007).

Hypothesising that the IPPCD failed to eliminate market failures, the policy can either have had no effect on the cost-effectiveness of the implemented measures, or a decreasing effect. The influence on the cost-effectiveness of the policy is dependent on the stringency of the emission limit values. If the values are not very stringent, the limits could equal the cheapest abatement options. More likely, however, is that there will be a decrease in cost-effectiveness since the mandatory nature of the abatement measures the IPPCD imposes on firms irrespective of their cost causes a reduction in the permit price. ETS participants with relatively low abatement costs who are subject to stringent IPPC regulations will see less incentive to abate thus leading to an increase in the total abatement costs.

Therefore, it is hypothesised that:

(1) mainly the companies subject to the IPPC directive will implement measures to reduce their emissions,
(2) the implementation of these measures will put negative pressure on the market price of the ETS,
(3) as these measures will not be the most cost effective, the cost-efficiency of the whole ETS will decrease and,
(4) since more efficient abatement mixes exist, the potential abatement measures still to be implemented in non-IPPC companies are more cost-effective than measures implemented under the IPPCD.

These hypotheses will be assessed using the NOx emission trading scheme as empirical basis. The methods of how the hypothesis will be tested is described in the next section.

# 3

# Methods and data description

As noted in the theory section, many of the previous studies on emissions trading system policy interactions have focussed mainly on theoretical, ex-ante interactions rather than on empirical data. One of the reasons for this is the relative novelty of market-based emission-reduction measures. The fact that the Dutch NOx Emission Trading Scheme was active for several years before it was appealed in 2013, make it a very suitable choice for the study of vertical policy interactions.

In order to assess not only the presence of interactions but also the magnitude of their influence on the effect of NOx ETS, this study uses an ex-post approach combined with quantitative evaluation methods. Since there is a lack of studies of vertical PI where the ETS is the lower tier, we will perform a two instrument PI analysis and provide an extrapolation to more general categories within the discussion.

Taking the theory presented before, it was hypothesised that the IPPCD emission limits in the environmental permits of companies have limited the cost effectiveness of the ETS. In order to test this hypothesis, the cost-effectiveness of the implemented NOx abatement techniques within the NOx ETS were deduced and set against a scenario in which no IPPCD limits were present.

To do this, the following research steps are executed:

1. Evaluate the NOx implied emission factors (IEF) per installation of all NOx ETS participants and identify the installations with a reduction in the EF;
2. Assess what NOx abatement technique was installed in the installation to reach the abatement level;
3. Estimate the cost of the installed abatement techniques per tonne NOx abated, deriving from various sources;
4. Analyse if the marginal abatement costs in IPPC installations were higher than non-IPPC installations and construct abatement potential without IPPC limits.

The NOx implied emission factors (IEF) per installation of all NOx ETS participants were evaluated and the installations with a reduction in the EF identified. The NOx abatement technique installed in the installation was assessed to arrive at the abatement level. An estimate was made of the cost of the abatement techniques per tonne NOx abated, derived from diverse sources. Whether the marginal abatement costs in IPPC installations were higher than non-IPPC installations was investigated and the abatement potential without IPPC limits constructed.

These research steps were used to assess what the cost-effectiveness has been of the retrofit of NOx abatement techniques compared to the least cost options in a counterfactual scenario. Only the costs of retrofit were assessed. The NOx abatement costs of newly constructed installations were not assessed since their abatement costs are inherently lower than retrofit costs and a meaningful comparison would not be possible. Moreover, NOx abatement costs have very limited influence on the investment decision to commission the construction of new installations.

## 3.1 Data description

The data used to evaluate industrial installations that have implemented $NO_x$ abatement techniques was collected by the Netherlands Emission Authority (NEa). The data of both the Netherlands NOx and EU $CO_2$ ETS the emission trading schemes was acquired by courtesy of the Netherlands Emission Authority. The data included the annual emission reports (emissieverslagen) of most participating installations (inrichtingen) in either pdf, word or excel format. The reports of the verifiers of the reports and other items such as e-mail conversations and additional data requests were also contained in the large data bundle.

An aggregated database provided with the data included the emissions and emission rights of all individual installations participating in either the $CO_2$, $NO_x$ or both emission trading schemes. The sheet also provided some information of fuel type, thermal capacity in $MW_{th}$ of the installation and whether the installation was a process or combustion installation. The initial database included a total of 2786 individual installations within different sites. The database was kept up to date through years 2006 to 2012. 2012 and 2013 were not or only sparsely included in the database as the decommissioning of the NOx trading scheme meant the file was no longer updated. The time-series was also incomplete and many hiatuses were found in the database. Since the database was a secondary source previously processed by the NEA, it was determined that several errors were present in the file and that not all hiatuses indicated that the installation was switched off.

To resolve the hiatuses, the primary data, i.e. the environmental reports, were used to amend the dataset. The primary emission reports, however, were not consistently reported; the style of the report, the formatting and the level of detail differed from year to year as no standard format was mandated prior to 2013 (R. de Ridder, Personal Communications). Even after trading commenced in 2005, firms were still able to cluster multiple installations that shared a single chimney. Where this was the case, the comparability was increased by clustering the installations in the database for the years that the installations were entered individually. One sector for which multiple sites could not be entered in the database was the refinery sector. For these sites, only the total emission rights of the site were given, and not per installation. As a result, the fuel input could not be calculated and thus neither the implied emission factor.

**Definitions**

**Site**: an activity on a commercial scale which is active within certain boundaries and is included in annex 1 of the *Besuit Omgevingsrecht*. When referred to in the text as company, only the activity of the company on that location is intended (*inrichting)*. A site often contains multiple installations.

**Installation**: individual unit within a site which can either by monitored individually or in clusters (*installatie*)**.**

**Process emissions:** emissions originating from the combustion of fuels to produce a product with a process PSR. Not to be confused with process installation.

**Combustion emissions:** emissions originating from the combustion of fuels without process PSR.

**Process installation**: installation within a process setting, being other than a boiler or turbine.

For a limited number of installations, the database also included an entry whether the emissions concerned combustion or process emissions. These entries, however, were not included for all installations and for at least a few were shown to be incorrect. To assess the type of emissions rights given to the installations, the primary sources were, therefore, used. Only industrial installations were investigated and, of those, only the installations of which it was anticipated process emissions would be possible. This included chemical and mineral producing installations and excluded all agricultural, paper and pulp and electricity, heat or steam producing installations. The investigations revealed that the emissions of the majority of the installations were combustion-related (2197); only 107 installations concerned process emissions. As the database also included sites which were participants of the $CO_2$ emission scheme, but not the NOx ETS, these installations were excluded. This mainly included smaller agricultural installations, packaging-producing plants and companies in the clay bricks industry. 302 installations were thus omitted reducing the total number of installations to 2484.

As a first step in researching the case, the database was processed to remove a large number of the errors. This was done by finding outliers in the implied emission factors and checking these with the primary data source. The years 2012 and 2013 were relatively important years for the trading scheme as in these years the emission rights were (almost) equal to the emissions. For this reason, the year 2012 was entered in the database manually. Due to lack of time, 2013 was excluded.

As the database only gives the emission rights and actual annual emissions, the fuel input was added to allow the emission factors to be derived. While the emission rights for process and combustion emissions are equal commodities within the trading scheme, the way the rights are earned differ. Combustion emissions are earned per the fuel input of installations, with installations gaining additional rights with increasing fuel input. Process emissions are earned by production in tons of product output. The following products are eligible for a process PSR:

**Table 1**: PSR sectors in the NOx ETS

| | | | |
|---|---|---|---|
| Iron and steel | Anodes | Stone Wool | Cement |
| Electrosteel | Magnesium oxides | Nitric Acid | Silicon Carbide |
| Aluminium | Active Carbon | Caprolactam | Nitrite |
| Zink | Phosphates:<br>-Phosphorus<br>-Phosphoric acid<br>-Tripolyphosphate | Enamelling:<br>-Continuous ovens<br>-Drum ovens | Glass:<br>-Flat glass<br>-Packaging glass<br>-Special glass |
| Carbon Black | | | |

Combustion emission rights are calculated simply by multiplying the fuel input by the annual PSR for combustion rights. To calculate the emission rights for process emissions, however, the relevant process PSR must be multiplied by the product output. A production location can either report only the process emissions in which all emission rights are calculated through a single PSR, or both process emissions and combustion emissions can be reported. In the latter case, the emission rights of each individual installations are calculated through the relevant PSR, being either the combustion or process PSR. This is often the case for sites where different products are produced, for instance in the chemical and glass producing industries where the emission report will state boilers, engines and turbines separately from process installations. As the emission factors for process emissions can thus only by calculated as grams NOx per tons of product output, the implied emission factors of the combustion and process emissions are discussed separately.

Since the emission rights for combustion emissions are based on the fuel input, the fuel input can be calculated using formula 4 and 5 below.

$$Emission\ rights_{combustion} = Fuel\ input_{\ in\ GJ} * PSR \tag{4}$$

$$Fuel\ input_{\ in\ GJ} = \frac{Emission\ rights_{combustion}}{PSR} \tag{5}$$

$$Emission\ rights_{process} = tons\ produ\quad * PSR \tag{6}$$

The calculated fuel input for several installation was checked against the reported fuel input in the emission report. This cross-reference showed the calculation to be highly accurate with only a very small error (less than 1%). This method, however, cannot be used for the installations with process emissions because the emission rights of these installations are calculated through the companies' process output as kilograms of NOx per tonnes of output and not the fuel input, as shown in formula 6. It was not possible to derive accurate fuel data for process installations on individual installation level as these were not reported in the emission report or in the electronic environmental annual reports (e-MJV). To assess the relative emissions of process installations, the emissions of the installation was divided by tons of product output. The output data per tons of product was, therefore, added to the database manually from the emission reports.

The reporting style of sites with a process PSR was heterogeneous, however. While for some sites the output data was reported in clusters of installations, or even for the whole site, for other sites the output was reported per installation. Moreover, some sites with a process PSR also contain additional combustion-emitting engines, boilers and turbines. Due to the heterogeneity of the process sites, the results of these sites are presented and discussed separately.

To derive the emission reductions of process installations, the implied emission factors for the process installation are calculated either by the emissions of a process installation times the output of that installation or by dividing the sum of emissions of all process installations by the total site output, depending on how the output was registered.

In addition to the above confidential disaggregated data, a non-confidential dataset was used on a site level in which the emissions and emission rights of companies were aggregated. The datasheet “NOx –emissiecijfers 2005-2013 per bedrijf”(NEa, 2014) is available from the NEa website

## 3.2 Selection of past implemented measures

To assess the cost-effectiveness of the implemented NOx emission-reduction measures, the database was used to find the sites and specific installations where abatement measures had been implemented. Because of large number of installations in the database, it was not feasible to assess the significance of the emission-reductions of every single installation. Furthermore, not every reduction in an emission factor of an installation points towards the implementation of NOx reducing techniques. Variability in emissions also occurs due to variations in production, an increase or decrease in the load of the installation or external factors such as outside temperature (European Commission, 2016). To counter this, a selection was made of sites and installations where there was a high probability that NOx abatement techniques had been implemented.

The selection of sites was done on the basis of reductions in the implied total-site implied emission factors and installation-specific implied emission factors. Assessing the emission reductions solely on a company level would not have been sufficient as companies which might have implemented a reduction technique on a boiler could have increased the use of diesel-powered back-up generators to achieve net zero or negative emission reductions. However, only assessing the emission reductions on an installation level would miss reductions that were due to the decommissioning of more polluting installations and, moreover, would have given a very high number of installations to assess.

The cut-off point for the selection was based on the reduction potentials of the various NOx abatement techniques as assessed by means of the literature study. After selecting the companies and installations with significant emission reductions, an assessment was made to find the implemented abatement technique. This was done using the following methods, decreasing in order of preference:

- Evaluation of the environmental report or public records;
- Direct contact with the specific company;
- Through expert judgment based on fit with other emission reductions in a specific industry.

Under the ETS, sites were obliged to notify the NEa of any adjustments to the monitoring plans and to include such plans in the annual emission reports. However, these adjustments mainly concerned adjustments to the methodology used in measuring the emissions and fuel use, and only to a lesser extent adjustments in installations or flue gas cleaning. Sites were not obliged to explain the reductions in NOx emissions per installation, though variability in emissions did contribute to a higher chance the site would be audited (R. de Ridder, Personal Communications). Moreover, the monitoring plan would not have been adjusted if, for example, a gas turbine already including hardware for steam injection would increase the use of injection, even though the NOx emissions would likely decrease. Therefore, where possible, the first method was augmented with data collected from the companies. Where companies were unwilling to participate, expert judgment was used.

### 3.2.1 Selection of cut-off point

NOx abatement techniques are primarily divided into primary and secondary measures. Primary measures entail adjustments to the combustion system which reduce the formation of NOx in-situ. Secondary measures are end-of-pipe techniques where the formed NOx is broken down after it has been formed (European Commission, 2016). The NOx abatement techniques are specific to the type of installation and the type of fuel.

The NEa database mainly contained combustion-type installations and had only a relatively small number of process-related installations. The combustion installations consisted of larger- and smaller-sized plants entirely focussed on the production of heat and electricity and industrial boilers, combined heat and power plants (CHPs), furnaces and other industrial installations. The main large fired power plants consisted of coal-fired power plants, mainly larger than 500 MW, and large gas-fired power plants in a CCGT or cogeneration set-up (where the latter also has additional firing in a boiler to generate steam in addition to the gas turbine). The industrial installations consisted of a large range of various sized boilers, single-cycle or combined-cycle gas turbines and heaters. Further process-oriented units were installations such as steam crackers, driers and other industry specific installations. A large number of the industrial installations were natural gas-fired, while some installations were fired with other industrial gasses. The abatement techniques of installations with a process PSR are discussed individually.

Given that a large source of NOx formation is the nitrogen content of the air (thermal NOx), primary NOx reduction techniques mainly focussed on regulating the air inflow in the combustion areas and reducing the temperature of the flame. These techniques were either retrofitted in existing installations or implemented in new boilers.

NOx emissions are also related to the nitrogen content of the fuel, being higher for coal and refinery gasses and negligible for natural gas, and the combustion temperature of the fuel. NOx emissions for hydrogen are, therefore, relatively higher as the combustion temperature is higher (European Commission, 2010).

For boilers, the BREF LCP (European Commission, 2006) differentiates between adjustments to the existing combustion systems and the implementation of low NOx burners (LNB). The former measures include configurations such as reducing the overall excess air in the boiler, forming multiple combustion zones in which oxygen is reduced in the primary combustion zones and increased in the secondary zone, and the implementation of flue-gas recirculation. Flue-gas recirculation reduces the amount of oxygen in the combustion air and therefore reduces the flame temperature. The instalment of low NOx burners changes the burner set up and changes the temperature and shape of the flame within the combustion system. The LNBs ensure a cooler, fuel rich flame with a lower oxygen content which ensures lower NOx formation. LNBs work with a more staged air inlet, ensuring a primary and a secondary flame to minimize the formation of NOx. The BREF LCP identifies three different types of LNBs, these being air-staged, fuel-staged and LNBs with flue-gas recirculation, while stating that the most-modern LNBs are a hybrid of all three types. The general NOx reduction rates given in the BREF LCP for primary measures in boilers are given in table 2 below (European Commission, 2006).

**Table 2**: reduction rate of various primary techniques in boiler installations (European Commission, 2006)

| Technique | Reduction rate |
|---|---|
| Low excess air | 10-44% |
| Air staging in the furnace | 10-70% |
| Flue-gas recirculation | 20-50% |
| Reduced air preheat | 20-30% |
| Fuel staging | 50-60% |
| Air-staged LNB | 25-35% |
| Flue-gas recirculation LNB | Up to 20% |
| Fuel-staged LNB | 50-60% |

The BREF LCP provides several notes on the reduction potentials of the primary measures, however. The first noted is that the range for the measures are relatively large. The potential differs strongly per installation and neither can all measures be installed within all installations. Moreover, the measure reduction rates can't simply be added or multiplied, but different measures can be combined. Especially the implementation of LNBs are frequently combined with the implementation of other primary measures. Since newly built plants already have these measures installed however, the additional reductions are mainly relevant for older plants with conventional burners or old-type LNBs. The ranges within the above table fit within the range of a BAT study of the Flemish Institute of Technological Research Vito, which places the range of effectiveness of primary measures between 10-50%, Only air-staging in the furnace falls outside this range (Goovaerts, Luyckx, Vercaemst, De Meyer, & Dijkmans, 2002).

For gas turbines, applicable primary techniques consist either of the injection of steam or water, or a mix thereof, into the turbine or the installation of dry low NOx emission systems (DLE). The high purity water or injected steam can attain emission reductions of 60 to 80% (European Commission, 2016). The measure can be retrofitted to existing turbines and implemented in new turbines. Adverse effects of steam or water injection are that the thermal efficiency of the turbine decreases, the fuel consumption increases and the stress to the material rises due to temperature shock. The reduction potential of the steam/ water injection can vary due to reduced loads; injection is only possible after a certain load has been achieved.

According to the 2006 BREF LCP (European Commission, 2006), there is a high degree of utilization of dry-low NOx emission (DLE) systems. According to the 2016 draft BREF LCP, the utilization of these systems can increase energy efficiency and electrical performance while also reducing NOx emissions (European Commission, 2016).

Secondary abatement techniques are measures installed to remove the NOx from the flue gasses after leaving the boiler or turbine and before leaving the stack. These so-called end-of-pipe techniques are separated into selective catalytic reduction (SCR) and selective non-catalytic reduction (SNCR) (Sorrels, Randall, Fry, & Schaffner, 2015; Sorrels, Randall, Schaffner, & Fry, 2015). Both techniques use a reagent such as ammonia or urea to reduce NO and $NO_2$ to $N_2$ (nitrogen). The SCR differs from the SNCR in that it uses a catalytic process which is placed down-stream of injecting the reagent into the flue gas stream. For the SCR system, different catalytic materials are used and different configurations are applicable depending on where in the flue-gas cleaning process the SCR is placed (either directly after the boiler, after the dust removal or at the tail-end after the desulphurisation) (European Commission, 2006). The SCR process can remove more that 90% of the NOx from the flue gas. The SNCR has a lower removal rate due to the lack of a catalyst. With the SNCR, the reagent is injected in the gasses after the boiler. As the reagent has a relatively limited temperature window for the reaction, the SNCR's effectiveness is dependent on constant boiler operation with few load fluctuations (European Commission, 2006). Whilst the SNCR is a relatively easy system to install, it has the disadvantage of having considerably lower removal potential than the SCR, being between the 30-50%. This means SNCR cannot be used to reach stringent NOx regulations for installations with very high NOx emissions without any other primary reduction techniques. The SNCR also cannot be used on gas turbines due to the longer residence time required, nor will it be very effective in installations with strong load fluctuations.

## 3.2.2 General cost functions

The cost of the implemented techniques deduced from the environmental reports, the company surveys or expert judgement, are found through literature sources. As the companies participating in the research are not asked to provide cost figures, the costs are estimated through more generic sources. An issue with the cost-estimation of NOx abatement techniques was that the complexity of the techniques differed from site to site but the information gathered from companies only provided limited information on the type of technique implemented. Because of this, the expected costs of the techniques should be regarded as a measure of comparison between the different techniques and for the scenario building.

The cost estimation was subdivided per fuel type, burner type and size of the installation. The main fuel types of installations with NOx abatement were coal and natural gas. The coal-fired units were all large combustion plants (LCP), the gas-fired units were subdivided into gas boilers, gas turbines and process furnaces (ovens, driers, etc.) and sub-subdivided into LCP (>50MW) and small and medium combustion installations.

To compare the marginal costs of the various techniques, both the annual capital costs and the operation and maintenance costs needed to be taken into account. To take account of the capital costs, a capital recovery factor (CRF) was used and multiplied with the capital investment. For the general cost function, adapted from Isaksson (2005), the annual cost of abatement as a function of total invested capital (K) was calculated with the long-term interest rate i, and the annual operation and maintenance cost of installation j in year t. The operator in brackets is the CRF, which also used the lifetime N of the technique. The cost of capital for utilities in Europe was used as long-term interest rate in the function. The value 6.59% was used as derived from Damodaran (2016) which was based on a large dataset of European utilities. The function used is presented in formula 7

$$C_{jt} = K_{jt}\left(\frac{i(1+i)^N}{(1+i)^N-1}\right) + O\&M_{jt} \qquad (7)$$

To derive the marginal or specific abatement cost per kg NOx abated, the annual costs are divided by the percentage reduction in implied emission factor (ΔIEF) multiplied by the fuel input in the latest operation year, as seen in function 8.

$$MC_j = \frac{C_{jt}}{\Delta IEF * F_j} \qquad (8)$$

## 3.2.3 Costs for coal fired power plants

The main additionally implemented technique for coal plants is found to be SCR installations. From an electric power plant database, it shows that before the start of the NOx emission trading scheme, the coal fired power plants are already equipped with primary NOx measures (PLATTS, 2012). This leaves an SCR instalment as the most viable option to abate NOx emissions.

The costs of an SCR installation is highly variable over different sites and installation (European Commission, 2016; EGTEI, 2014). The total costs of an SCR depends on both the investment costs in the SCR installation and an annual operation and maintenance costs (O&M) existing of the costs of the reagent, electricity and catalyst (replacement) costs (EGTEI, 2014).

Due to the variability in cost estimations, the latest BREFF for Large Combustion Plants (LCP) does not give cost estimations for implemented SCRs in coal fired power plants, but does give some cost estimations for gas turbines in a CCGT set-up (European Commission, 2016). The US Environmental Protection Agency (EPA) (Cichanowicz, 2010) does provide more up-to-date figures on the costs of SCRs in various installations, including large coal fired utility boilers. Cost estimations for gas fired electricity producing installations are not given by the US EPA, but data on gas fired industrial boilers is provided.

The US EPA (Cichanowicz, 2010) shows the costs of an SCR on a coal-fired power plant to have a spread of 100-275 $/kW in 2008 dollars. A slight economy of scale was present, where the cost for a 200 MW unit was 200 $/kW, and 160 $/kW for a 800 MW unit. The average cost for SCR in the TFTEI "Estimation of costs of reduction techniques for LCP" cost calculation, shows the average cost of 11 installed SCRs to be 122.84 EUR/KW, with a range of 81-138 EUR/KWe.

Because the various cost examples from the US EPA, the BREF documents and publications from Vito and EGTEI show a broad range of cost data, the US EPA cost manual was used (EGTEI, 2014; Goovaerts,Luyckx, Vercaemst., De Meyer. en Dijkmans, 2002; Sorrels, Randall, Schaffner, et al., 2015). The aim of the US EPA cost manuals is to provide cost-estimations for study-level purposes; the cost estimations are based on previous cost estimations made by the US EPA and fitted to generic function form (US Environmental Protection Agency, 2002). It was recognized that the actual costs would deviate from the calculated costs, but that they would be in the same order of magnitude.

The cost function derived from the new US EPA draft chapter in Sorrels et al. (2015b) was used. This function bases the total costs of the SCR on three elements, these being: the initial SCR costs (formula 10), the cost for the reagent preparation (RPC, formula 11) and other costs included in the balance of plant costs (formula 12). These costs are multiplied by a factor of 1.3 to take the retrofit costs into account.

Since one of the largest sources of imported coal in the Netherlands has a relatively low sulphur content (Schmidt, 2008), it was assumed that the power plants use relatively low sulphur coal. This eliminated the need for an air preheater.

**Definitions and variables**

**MWth:** thermal capacity of the installation in Megawatt

**MWe:** electrical capacity of the installation in Megawatt

**NRF:** NOx removal factor in lb/MMBtu, normalised factor of NOx removal for SCR. The NRF is 1 at 80% efficiency, >1 at higher removal efficiencies.

**NPHR**: Net productive heat rate. Inverse of efficiency with the units Mmbtu/ per MWh.

**HRF**: heat rate factor, normalised factor of NPHR. Equal to NPHR/10.

**CoalF:** coal factor depending on type of coal

**TCI:** Total cost of Investment

**PBC:** Balance of plant cost

**RPC:** Reagent preparation cost

The functions all used a constant factor provided in Sorrels et al. (2015b) and several parameters including the NOx removal factor (NRF), coal factor (CoalF), boiler electrical capacity (BMwe) and the heat rate factor (HRF) and the net productive heat rate (NPHR); see the textbox above.

The NOx removal factor is a normalised factor of the NOx removal efficiency, the NRF is normalized at 80%. Installations with a removal efficiency higher or lower than this value thus have a NRF over or below 1 (Sorrels et al., (2015b). The NOx removal efficiency is calculated by means of the collected emission data, where the initial high NOx emission factor was used as a proxy for the NOx levels before implementation of the measure. As the function uses the unit lb/MMBtu, the emission factors of gram/GJ were converted by a factor of 0.0125.

The net productive heat rate (NPHR) is the inverse of electrical efficiency and has the unit of MMBtu per Megawatt-hour. Both the MMBtu fuel input and Megawatt-hour electrical output were derived from data from the electronic environmental reports (e-MJV). The boiler MWe capacity was derived from the online sources and the COMPETES model. The heat rate factor (HRF) was calculated by dividing the NPHR by the typical NPHR value 10 to provide a normalization (Sorrels et al., 2015b).

The coal factor is dependent on the coal quality used. The main source countries for imported coal in the Netherlands are Colombia and South Africa (Greenpeace, 2008). As the main type of coal in Colombia is bituminous coal which is also the main exported type of South African coal, the coal factor for bituminous coal was used, (Schmidt, 2008; Tewalt et al., 2006). As the coal factor for bituminous coal is 1, it drops out of the equation (Sorrels et al., (2015b). The various factors were scaled by the fixed scaling factors given in Sorrels et al. (2015b). Lastly, the costs were transferred from 2012 dollars to 2016 euro's, this was done by dividing the calculated costs in dollars by the approximate exchange rate of 1.2 dollars per euro and multiplying them by an assumed inflation rate of 4% to the power 4.The functions used for these calculations are shown below

$$SCR_{SCR\ coal} = 270.000 * (NRF)^{0.2} * (\ B_{MWe} * HRF)^{0.92} \tag{9}$$

$$RPC_{SCR\ oal} = 490.000 * ((NOx_{in} * B_{MWe} * NPHR) *\ \eta_{NOx})^{0.25}$$

$$BPC_{SCR\ coal} = 460.000 * (B_{MWe} * HRF)^{0.42} \tag{10}$$

Combining these three calculations, makes the Total Cost of Investment (TCI) for SCR units to be:

$$TCI_{SCR\ coal} = \left[\frac{(SCR_{\$\ cost} + RPC + BOC) * 1.3}{1.2\ \$/€}\right] * (1 + 0.04)^4 \tag{11}$$

| | | |
|---|---|---|
| $SCR_{\$\ cost}$ | = | SCR cost in US-$ |
| 270.000 | = | Constant in equation |
| NRF | = | NOx removal factor, fraction NOx removed*0.0125 |
| $B_{MW}$ | = | Boiler electrical capacity in MW |
| HRF | = | Heat rate factor, *0.1 |

The annual costs were then added to the total investment costs. The methodology suggested by Sorrels et al. (2015b) has a high data demand but this, however, is largely unknown. For example, to estimate the cost of the reagent and catalysts, data is needed on the volume of catalyst, volume of reagent flow per hour, lifetime of catalysts, number of catalyst reactivations as well as additional data. To reduce the data demand, an approximated costs of 0.6 cents per MWh was used as presented in Cichanowicz (2010) based on the O&M costs of a 500MW coal-fired utility boiler.

### 3.2.4 Costs for gas fired units

**Cost for SCR and SNCR**

From section 3.2, it showed that several NOx abatement options are available for gas fired units. The main installed primary measures are the instalment of low NOx burners (with or without overfire air) on boilers, wet injection or DLE on turbines. SCRs and SNCRs as generally employed secondary techniques.

The BREF LCP provides several cost estimations of SCRs for gas turbines, the specific abatement costs are estimated for gas SCR units in the range of 3500-6300 EUR/ tonne NOx for units smaller than 25MW and 1940 EUR/ tonne NOx for units of 170 MW.

The BREF for the refining sector (REF) (European Commission, 2010) states that a the costs for an SCR unit on a reformer furnace in 1998 was 3.2 million EUR for a 68 MW, which corresponds to an amount around 5-6 million EUR in 2016 euro depending on interest used. Two other SCR units are mentioned in the BREF REF, having investment costs of respectively 3.2 million and 2.2 million EUR in 1998, with annual costs between 1.1 and 0.5 million per year.

The cost estimation for the total cost of investment of gas fired units from Sorrels et al. (2015b) is only based on one function, provided in formula 14. The function scales the cost per kW of an SCR of a 200 MW installation in 2012 to the size of the unit of calculation. The annual O&M costs are assumed to be equal to the O&M costs but for industrial units the electrical MWh output is either not relevant or unknown. Therefore the average of the share of O&M costs in the total annual costs are calculated for the coal fired units and this average was then used to calculate the costs for the gas fired units as share of the capital recovery costs.

(12) $$Total\ cost\ of\ investment\ (TCI) = 80 * \left(\frac{200}{B_{MWe}}\right)^{0.35} * B_{MWe} * 1000\$/€$$

The costs for SNCR units for gas-fired installations over 25 MW were calculated using the formulae based on the functions of Sorrels et al. (2015a as shown in formula 15, 16 and 17. The total investment costs were based on the costs of the SNCR unit and the balance of plant costs, which include other costs such as piping, ductwork and auxiliary power modification (Sorrels et al, 2015a). The total investment costs included a factor 1.3 for the engineering, management and installation costs for retrofitting the installation. Both functions had a constant in the equation and were scaled accordingly. In the SNCR formula, the heat-rate factor was assumed to be 0.82 as stated in Sorrels et al. (2015a) as a default value for gas-fired units. The formula was adapted to account for the fact that the heat rates of the installations were unknown. The NOx removed/hr is calculated through dividing the annual removed NOx in kg by an estimated operating hours, 6000 hours/year is assumed as default value.

(13) $$TCI_{SNCR} = 1.3 * (SNCR_{cost} + BO\ \ _{Cost})$$

(14) $$SNCR_{Cost} = 147.000 * (B_{MWe} * 0.82)^{0.42}$$

(15) $$BOP_{Cost} = 213.000 * (B_{MWe})^{0.33} * (NOx\ removed/hr)^{0.12}$$

As noted for the SCR calculation, the calculation of O&M costs have a high data demand. To limit the use of assumptions, a more simple calculation was made. According to the ICAC SNCR Committee (2008), the annual O&M costs are around 15-35% of the total annual costs; the capital recovery costs constitute the other part. The O&M costs, therefore, make up a significant part of the total annual costs. For the SNCR calculations, it was assumed that the O&M costs would constitute 25% of the annual costs.

**Cost for LNB and ULNB**

DLE and LNB adaptations are both relatively cost-effective options for attaining NOx emission-reductions from gas-fired units. The BREF LCP (EC, 2016) states that the costs for DLE systems also show significant variations depending primarily on manufacturer and process type. The source of these cost variations related to differences in performance, design complexity and reliability factors.

An additional difficulty in estimating the costs of LNB and DLE is the variability in the cost of the units while, the estimation of the costs of specific LNBs is complicated by the probability the burners were installed towards the end of the lifetime of the older burners. From a provider of ultra LNB's, it was learnt that a burner of around 2-4 $MW_{th}$ cost around 7000 euro each. The cost of the burners, however, are only a minor part of the total investment costs. The 'other costs' mainly constitute engineering, material and construction costs.

Vito presents a range of costs of low NOx burners (see table 3), which revealed that larger installations were able achieve significantly lower investment costs as a result of increased economy of scale (Duerinck, Cornelis, & Rompaey, 2002). The cost function for all LNBs is however not as simple that a cost function integrating increasing costs to scale can estimate the costs well. As the costs of the burners is only a fraction of total costs, the costs are highly dependent on the ease of retrofit (US EPA, 1992). The costs of LNBs are thus dependent on the complexity of the instalment, this complexity increases when the additional space needed for the LNBs is not available. This discrepancy shows when comparing costs estimates from various sources. Kroon (2015) for instances shows the costs of a 50 MWth boiler being €87.000, the US EPA estimates these cost to be 346.000-955.000 in 1991 dollars and Seebold (2013) shows these costs to be $3 million for an installation of ULNBs in a 150 MMBtu /hr (44 MW) installation. The BREF for refineries provides an example where the cost of burners is 7000 EUR/burner, but that additional costs form 540% of burner cost. These cost ranges make clear that the LNB boiler cost estimation of Kroon (2015) is not applicable for process furnaces within, at least, the refinery and chemical sector. Therefore different cost functions are used for boiler installations and process installations. Process installations are defined as combustion installations being other than boilers, turbines and cogeneration units. These mainly are composed of (steam) crackers, furnaces and driers and are characterised by higher retrofit costs.

The burners that have been replaced, however, are likely to have reached the end of their lifetime as all burners installed after 1997 reached emission levels around 30-40 grams NOx/GJ (Kroon, 2015). These NOx abatement measure will thus have been very cost-effective since the additional cost of LNBs in respect of older generation boilers is low (EC, 2016). Kroon (2015) states that many of the burners which have not reached emission levels of 70 mg/ $Nm^3$ (20-25 g/GJ) are likely to be more than 20 years old.

It is thus assumed that burners can be retrofitted at high cost-effectiveness up to an emission level of 20-25 g/GJ with a reduction rate of 28-50%. The costs for these units are estimated to be around 1.7 €/ $kW_{th}$ as calculated using formula 18. With emission reductions over the 50% range, it is likely that a more advanced, ultra LNB is installed as seen in table 4. This ULNB results in higher abatement costs but also in increased reduction levels according to the BREF REF but also face higher costs(European Commission, 2010). Therefor a mark-up for ULNB is used.

$$C_{LNB\ boiler}\ kWth^{-1} = 986 + 1721 * B_{MWt} \quad (16)$$

$$C_{ULNB\ boiler}\ kWth^{-1} = C_{LNB} * 1.2 \quad (17)$$

**Table 3 and 4**: Cost of LNB as presented by Vito (Duerinck et al., 2002)

| Capacity in $MW_{th}$ | Investment in €/ $kW_{th}$ | O&M in cent/$kW_{th}$ |
|---|---|---|
| 1500 | 9 | 0.015 |
| 600 | 12 | 0.04 |
| 500 | 12 | 0.015 |
| 400 | 13 | 0.015 |
| 160 | 4-16 | 0.015-0.05 |
| 30 | 28 | 0.03 |
| 10 | 2 | |

| Burner type | Reduction rate | Annual costs in € |
|---|---|---|
| LNB | 28-50% | 2818 |
| Ultra LNB (first gen) | 55% | 4087 |
| Ultra LNB (last-gen) | 75-85% | 613-908 |

The cost examples of Seebold (2013) show a very large range at low capacity process installations and a linear trend ranging from 100 to 350 $ per MMBtu/hr. Seebold (2013) presents several costs examples for LNBs on refiner installations, being 2 million $ for a 15 MW installation, 3.2 million dollars for a 60 MW installation and over 4 million for a 100 MW installation. This translates to a respective 133, 53 and 40 €/ $kW_{th}$..

Converting the cost estimations in the BREF REF to 2016 euro's with a 2% annual inflation rate, provides cost examples per burner ranging from 100.000 euro in 1998, € 28000-30.000 in 2007,€ 44.000 in 2008 and € 60.000 in 2009, with an average of € 40.863 per burner. Assuming a 1 $MW_{th}$ burner size, this comes down to €40.863 /$MW_{th}$. This cost range is however still lower than the range presented in Seebold (2013), but as the BREF REF is tailored to the EU market, the cost estimations are based on this document. The function used is presented in formula 19.

(18)

$$C_{LNB\ Process} = 40.863 * B_{MW\ th}$$

To conclude, to calculate the costs for retrofitting LNBs to achieve emission levels up to 20 g/GJ within sectors other than the refinery industry, formula 18 as used by Kroon (2015) was applied. For emission levels lower than 20 g/GJ and reductions over 60%, it was assumed that an ULNB was installed and a 20% mark-up was applied as noted in formula 19. For the refinery and large-volume chemical industry, the costs were calculated using formula 20, to take account of higher costs due to technical engineering issues. Operation and maintenance costs were assumed to be zero. Even though the Vito publication (Duerinck et al., 2002) shows O&M costs, Kroon (2015) states that these costs are negligible and are therefore omitted.

**Costs for DLE**

The BREF LCP estimated the retrofitting costs of dry low NOx combustors to be 20-40 per kWe, which amounts to approximately 2 to 3 million EUR for a 140 $MW_{th}$ gas turbine (European Commission, 2006). An additional 500.000 euro per year was given as maintenance costs. The investment costs were specified as being approximately 15% higher and maintenance costs approximately 40% higher than non-DLE combustors. However, since DLE systems have a higher lifespan than systems with steam or water injections the additional capital investment is only 10% higher and O&M costs limited.

The cost estimations in the draft BREF LCP fit relatively well with the cost range of Duerinck et al. (2002), as presented in table 5. The BREF LCP specifies slightly higher costs for small installations and lower costs for larger installations than Duerinck et al. (2002). Kroon (2015) estimates the costs for DLE for installations up to 10 $MW_{th}$ to be 60 €/$kW_{th}$ and for installations over 20 $MW_{th}$ to be 30 €/$kW_{th}$, which is higher than both Duerinck et al. (2002) and the BREF LCP (2006 and 2016) estimates. However, Kroon's cost estimations only reveal marginal economies of scale in the O&M cost for DLE and the examples of Duerinck et al. (2002) don't show any clear economies of scale. It should be noted, however, that Kroon's costs are based on a 1999 publication which may cause an over-estimation relative to more recent costs.

In order to extrapolate the cost examples of the above publications to the implemented measures found in the research, the cost examples of Duerinck are used to make a general cost function. This is done by fitting a trend line through the data points of Duerinck et al. (2002), the data in table 5 is used as input data. A power function showed the best-fit, showing an $R^2$ of over 0.83 pointing to a good fit as depicted in graph 1. The cost per $kw_e$ were thus calculated using the function presented in formula 21 in which the electrical capacity is the dependent variable

Although data on the electrical capacity of all utility turbines is available from other sources (through PLATTS or the ECN COMPETES model), the electrical capacity for industrial turbines is not known. The thermal capacity was, therefore, used to calculate the electrical capacity of these units. A 25% reference efficiency was used for a gas turbine in a single cycle set-up, while a 40% efficiency was used for a combined cycle set-up where the steam turbine was included in the thermal capacity. Electrical capacities for CCGT units were not calculated since only those CCGT units were selected where the electrical capacity was known.

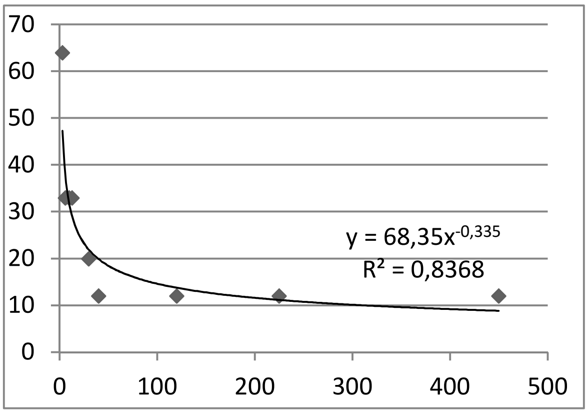
Variable costs were derived from Kroon (2015), the mean variable costs were posed as 0.53 €/MWth/hour, which, assuming a 40% electrical efficiency, corresponds to a 1.32€/MWe/hour. Full load hours were calculated by dividing the energy content of the fuel input by the thermal capacity of the unit. Since the installation is unlikely to run solely on 100% load, a factor 1.3 was used to account for lower loads and to convert full load hours to operating hours.

$$TCI_{DLE} = \left(68.35 * \ \mathrm{Cap_{kW-e}}^{-0.335}\right) * \mathrm{Cap_{kWe}} \tag{19}$$

$$O\&M_{DLE} = 0.53 * MWth * \left(\frac{fuel\ input * 0.277 \frac{GJ}{MWh}}{Cap_{MWth}}\right) * 1.3\ \frac{operating\ hour}{full\ load\ hour} \tag{20}$$

**Table 5**: cost estimates for DLE presented in Duerinck et al. (2002) **Graph 1**: cost extrapolation for DLE systems in turbines

| Capacity in $MW_e$ | Investment in €/ $kW_e$ | O&M in €/ year |
|---|---|---|
| 450 | 12 | 700.000 |
| 225 | 12 | 675.000 |
| 120 | 12 | 290.000 |
| 40 | 12 | |
| 30 | 20 | 36.000 |
| 13 | 33 | 29.000 |
| 10 | 33 | |
| 6 | 33 | 21.000 |
| 3 | 64 | |




**Cost for steam injection**

Steam injection is not regarded as best available technology and retrofits generally focus on the installation of DLE. The BREF LCP, however, does provide a cost example for the retrofit of a new steam injection system in a 3x 22 MW steam turbine, including a Heat Recovery Steam Generator (HRSG) (European Commission, 2006). The rebuild of the gas turbine was estimated to be 300 NOK million, with an additional 10 NOK million for the HRSG and a 6 NOK million for a de-salination plant for water production. The BREF REF (European Commission, 2010) provides a cost example of an 85 $MW_e$ plant with operating costs amounting to 0.8 million euro. A similar cost figure is given In the BREF LCP (EC, 2016) which estimates the costs of steam injection for a 140 MWth gas turbine to be EUR 1.7 million. These operating costs are mainly due to reduced efficiency caused by an increase in the energy consumption of the steam injection system, loss of turbine performance and increased stress on the turbine. Several cost estimations are given in the BREF REC for the variable costs of various sized gas turbines, with estimates of 0.4, 0.24 and 0.15 $/kWhr being cited for 4, 22.7 and 161 MW turbines respectively. The cost estimations for steam injection were derived from the above examples and linearly scaled to the specific thermal capacity.

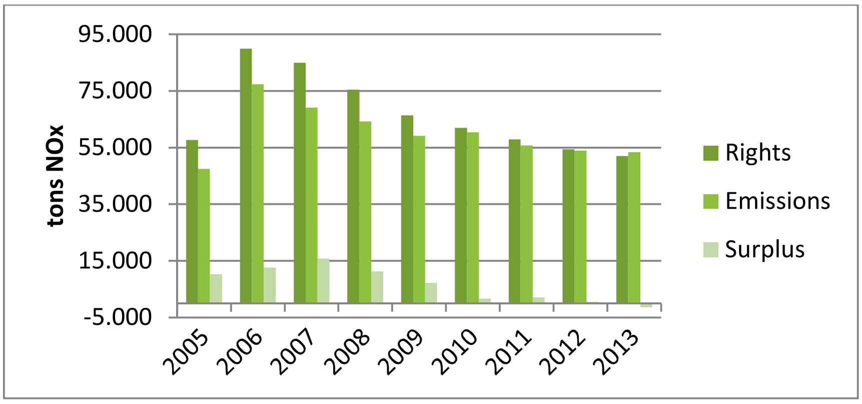
# 4 Results

This chapter presents the primary results of the analysis of the data provided by the NEa. The first section provides a general introduction to the emission-reductions in each of the ETS sectors. The following chapters present the results of each of the four steps used in the methodology and compares the cost-effectiveness of the implemented abatement techniques with the estimated emission reduction potential of 2006. Section 4.2 describes the selection of sites based on the emission factor reductions; section 4.3 gives an overview of the implemented abatement measures and section 4.4 details the cost-estimation. Finally, in section 4.5 the aggregated costs of the implemented abatement measures are compared to the cost-effective abatement potential remaining in the NOx emission trading scheme.

## 4.1 Introduction

The NOx ETS came into force in 2005 and ran through 2013. As graph 2[1] shows, each year following the introduction of the scheme saw a reduction both in the level of emissions and in the total number of available emission rights. Between 2006 and 2013, the total amount of emissions declined by over 31.9% while the number of issued emission rights declined by 42.2%. During the scheme's first five years of operation, the number of rights significantly surpassed the level of emissions, in 2008, however, the surplus started to decline and by 2013 there was a shortage of available emission rights. Although under the emission trading scheme, the amount of emissions declined, the number of companies participating in the scheme increased considerably from 259 in 2006 to 346 in 2013. However, in spite of the relatively large number of new market participants, the total volume of activity in the scheme, calculated as the total fuel input in combustion installations, remained relatively stable with a total fuel input increase of only 0.27%.

**Graph 2**: emissions, emission rights and net surplus of the NOx ETS



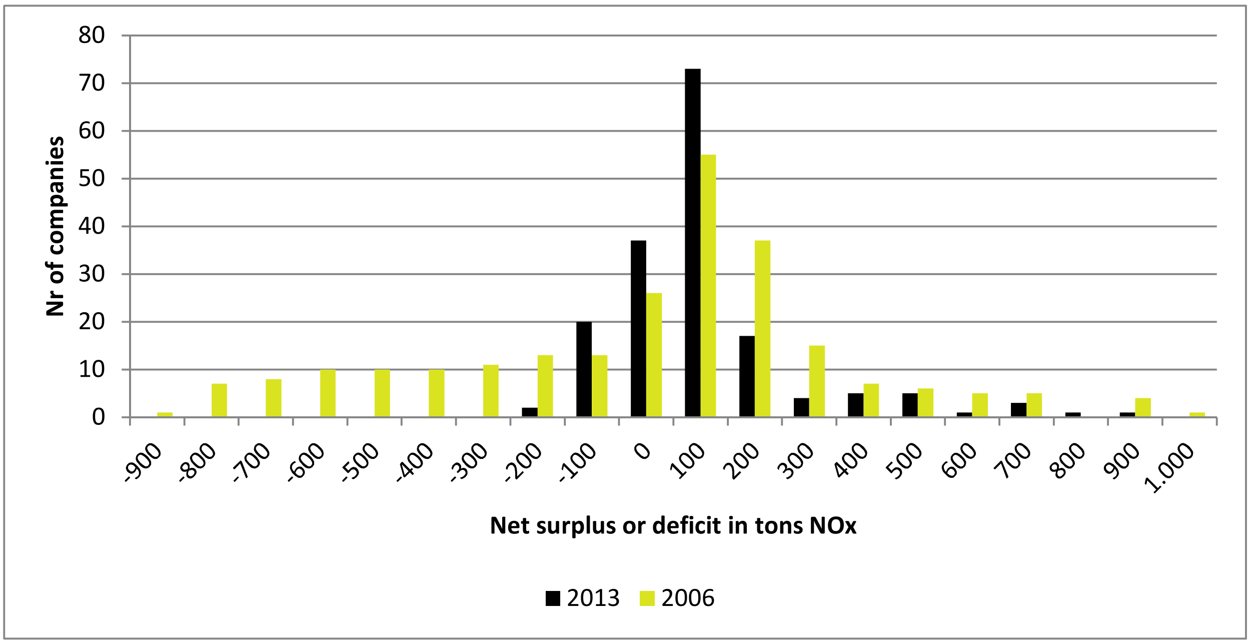

[1] In the graph, 2005 cannot be fully used as a representative year, as trading only commenced in June of that year.

In 2006, 61.5 ktons of the 77 kilotons (kton) NOx total ETS emissions originated from combustion sources only and 15.8 ktons NOx from sources that generated either only process emissions or both. By 2013, the total ETS emissions had decreased to 53.4 ktons, of which 39.6 ktons from combustion emissions and 13.85 ktons from installations with a process PSR. The share of the total ETS emissions from process PSR thus rose from 20.4% to 25.9%.

The NOx emission trading scheme was characterized by an uneven distribution of deficits and surpluses of emission rights (Nederlandse Emissieautoriteit (NEa), 2014) with some large companies having a substantial oversupply and others a sizable shortage. This unevenness held over the whole trading period although the distribution evened out towards the end of the trading scheme. As the distribution bar chart in graph 3 shows, however, the majority of firms had net zero emission right surpluses and only a few firms had a significant non-zero balance. The number of firms with a zero surplus or deficit of emission rights was higher in 2013 relative to 2006, but of the participating firms in 2013, two had a deficit of over 900 tons, two had a surplus of over 800 tons and five a surplus of over 700 tons NOx.

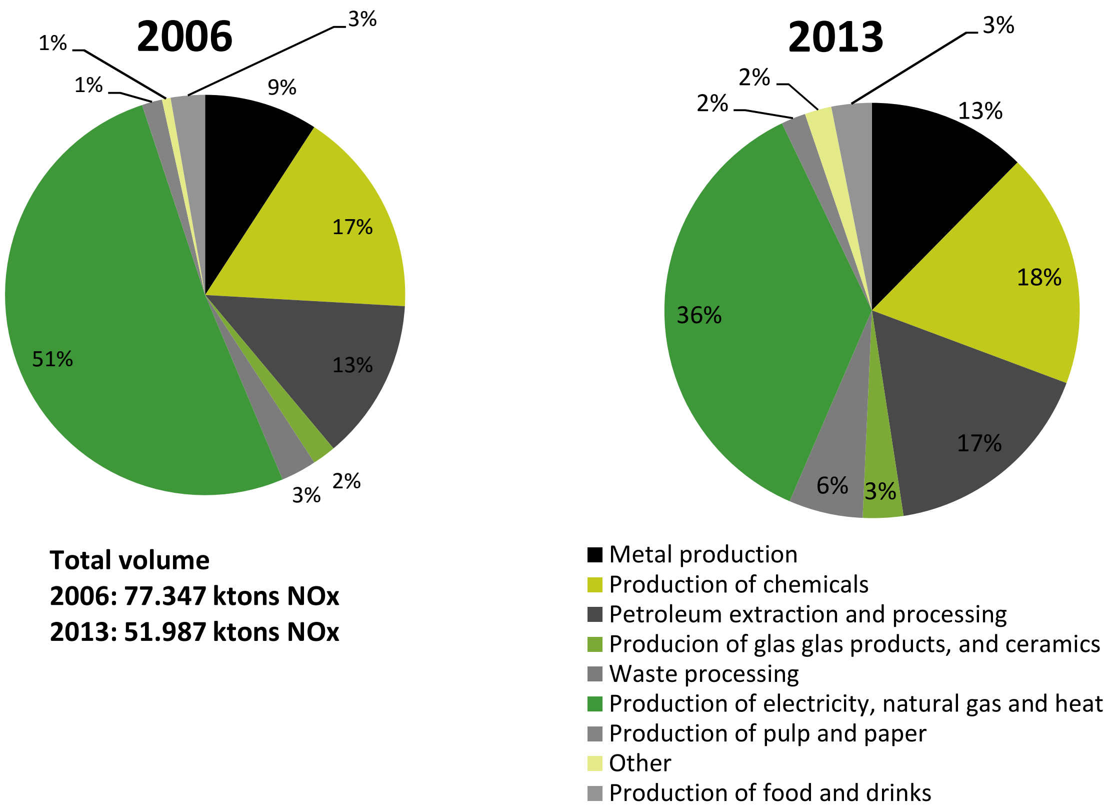
The ten participants with the largest deficits in 2006 were mainly large (coal) fired power plants, the Shell refinery and TATA Steel steelworks. In 2013, four of these firms were still in the top 10 but three had acquired a net surplus of emission rights, with two plants which in 2006 had had the largest deficit of rights acquiring the largest surplus. Of the firms that were in the surplus top 10 in 2006, four were still in the list in 2013. New entrants to the surplus top 10 in 2013 were mainly the newer power plants such as Magnum and Sloe and the newer installations in the Maxima power plant.

**Graph 3**: Distribution of surpluses or deficits of emission rights of participants of the NOx ETS




With regards to the sectors that account for the highest proportion of NOx emissions, as the pie charts in figure 4 and 5 show, the production of electricity, natural gas and heat sector (short: electricity or power sector) was the principle source of emissions under the ETS. Between 2006 and 2013, emissions from this sector did decrease significantly from 51% of the total in 2006 to 36% in 2013 making it also the largest source of NOx reduction. During the same period, the total emissions from all other industrial sectors only decreased by 10.9% while in some sectors, for example in the metal, chemical and petroleum producing industries, their share of NOx emissions even increased. However, although the relative shares in total emissions increased for these sectors, the absolute emissions decreased by a respective 8.1%, 25.3% and 11.5%. Several smaller industries increased their absolute emissions, including the glass and waste processing industries (14% and 37.3% respectively).

**Graph 4 and 5:** pie charts of the relative share of the emissions of the main industries in the NOx emission scheme.




Reduction in total sectoral emissions can be decomposed in three effects, which incorporates *volume* effects, *structural* effects and the emission *intensity* effect (Xu & Dietzenbacher, 2014) Volume effects refer to the total emissions of a sector or site which increase or decrease as the volume of fuel input increases or decreases. The structure effect refers to a change in the structure of technologies, in this case being the mix of emitting installations used. Examples of the structure effect would be if a new entrant with low emissions was to enter the scheme and as a result decrease the sectoral emission relative to the fuel input, or if the usage of a more polluting installation was increased in a site and cause the site emission factor to increase. The emission intensity effect refers to a reduction of the emission factor of an installation and, therefore, of the site or sectoral total emission factor. The latter can be caused by NOx abatement techniques, but also by, for example, a different load or improved air regulation of the installation.

The implied emission factors for 2006 and 2012 of the participating sectors with combustion emissions including the difference between the two (delta, Δ), are presented in table 6 below[2]. The table shows that mainly the electricity sector and the petroleum processing sector achieved reductions in the implied emission factor. Some other sectors achieved smaller reductions, while still others did not attain any reductions but increased their implied emission factor.

[2] The emission factors in the table are solely calculated for the participants with combustion emissions. The emission factors of the process emissions cannot be aggregated to sectoral averages due to large differences in reporting of the emissions. The table also shows the emission factor for the metal production, even though most of these emissions are process emissions. The participants taken up in the table only have combustion emissions and include for instance companies which produces secondary metals out of scraps.

**Table 6**: Implied emission factor for combustion emissions per sector

| Sector | IEF 2006 gr NOx/ $GJi_{input}$ | IEF 2012 | Δ 2006-2012 |
|---|---|---|---|
| Petroleum and natural gas extraction | 119.3 | 114.2 | -4% |
| Petroleum processing | 57.4 | 37.2 | -35% |
| Waste processing | 34.8 | 38.5 | 11% |
| Agri and horticulture | 56.6 | 58.4 | 3% |
| Asphalt production | 80.4[3] | 72.4 | -10% |
| Production and distribution of electricity, heat and natural gas | 62.0 | 32.5 | -47% |
| Loading, unloading and handling and storage | 32.1 | 44.7 | 39% |
| Metal production | 73.9 | 68.6 | -7% |
| Production of chemical products | 38.9 | 36.5 | -6% |
| Production of pulp, paper and cardboard | 49.0 | 46.9 | -4% |
| Production of food and drinks | 48.7 | 44.5 | -9% |
| Other | 68.3 | 80.8 | 18% |

The decrease or increase in emission factors per sector can be used to calculate to what extent total emissions within the ETS were caused by reduced activity or by lower relative emissions in the various sectors. The total ETS NOx emission reduction was 24.4 ktons of which 21.7 ktons was due to a reduction in sites with wholly combustion sources. Using the emission factors of 2006 and the activity levels of 2012 to calculate a frozen relative emission baseline, the combustion emissions in 2012 would have been 60.1 ktons NOx instead of 39.8 ktons NOx. Within this frozen emission baseline, the 2006 emission factors per industry was multiplied by the total energy input of 2012 to calculate the level of the emissions if the emission factor had remained constant. As the fuel input within the whole ETS remained relatively the same, the emissions would have been almost equal to 2006 if the emission factor hadn't changed.

The effect of the frozen baseline per sector is presented in table 7 below, which compares the actual absolute emissions per sector with the frozen baseline and shows the difference between the two. Sectors where the emission in the frozen baseline are higher than the actual emissions in 2012 have decreased their emission factor and have, therefore, either achieved emission reductions through the emission efficiency effect or through structural effects. Sectors with a negative delta increased their emission factor between 2006 and 2012 either due to a more polluting process or through the addition of new, more polluting companies.

The table shows that a reduction of over 20 ktons NOx was achieved due to reductions in the emission factor between 2006 and 2012. As the increase in fuel input was relatively limited, it can be concluded that the volume effect was low. The reduction of the emission was for some industries significant, mainly for power plants and the petroleum processing sector. The table shows that the power-generating sector had the greatest influence on the reduction of total emissions due to a lower emission factor, which amounted to an almost 17 ktons NOx reduction compared to the baseline. Alongside the electricity sector, the petroleum refinery sector attained an emission reduction of approximately 3 ktons. The remaining industries had relatively little influence while several sectors increased their emissions.

[3] The asphalt sector only joined the ETS in 2008, therefore the 2008 value is used in the 2006 column for this sector.

**Table 7**: Actual emissions and emissions calculated through the frozen emission baseline.

| Actual emissions | Actual emissions 2012 | Frozen baseline 2012 | Δ Emission reduction |
|---|---|---|---|
| Petroleum and natural gas extraction | 2,961 | 3,095 | 133 |
| Petroleum processing | 5,437 | 8,392 | 2,955 |
| Waste processing | 2,928 | 2,649 | -279 |
| Agri and horticulture | 760 | 769 | 9 |
| Asphalt production | 128 | 69 | -59 |
| Production and distribution of electricity, heat and natural gas | 18,697 | 35,691 | 16,994 |
| Loading, unloading and handling and storage | 38 | 27 | -11 |
| Metal production | 34 | 37 | 3 |
| Production of chemical products | 331 | 280 | -51 |
| Production of pulp, paper and cardboard | 5,731 | 6,111 | 379 |
| Production of food and drinks | 1,045 | 1,093 | 48 |
| Other | 1,722 | 1,885 | 163 |
| Sum | 39,812 | 60,096 | 20,284 |

The decrease in sectoral emission factors, can either point to structural effects or emission intensity effects. One aspect of the structural effects is that polluters with either high relative emissions exited the scheme or polluters with low emissions entered the scheme[4]. To assess whether these structural effect were significant, the change in emission factor was calculated by removing all firms that entered or exited the scheme after 2005. This brought the number of participating firms down from 363 to 180 and caused a decrease in the total emissions in 2012 from 39.8 ktons to 33.9 ktons. The influence was less on the 2006 emissions which only decreased from 61.5 ktons to 59.7 ktons NOx.  This points to more participants having entered the scheme than had exited; had more participants active form 2005 have exited the scheme, the emissions would have shown to decrease further.

The effect of removing these companies from the emission factor calculations as described above, was positive for several smaller industries. For the main industries, including power, refinery and chemical industry, the effect was marginal. For example, the emission factor for the electricity sector would have been 4% higher should all entries and exits had been removed. This means that the emission factor was 4% lower due to new companies entering the scheme. Since removing these companies caused only a 2% decrease in the 2012 emission factor, the effect on the scheme as a whole was rather negligible.

This points to the emission intensity effect being the main cause in the reduction in the total emissions of, mainly, the electricity sector and the petroleum processing sector, and that therefore it is likely that significant emission abatement techniques have been implemented.

[4] The other aspect of structural effects would be the structure of production within firms, i.e. the implementation of various installations. This effect can't be decomposed on this sectoral level as disaggregated activity within emission sites would be needed.

## 4.2 Selection of sites

To determine which firms in the sectors discussed in the introduction actually implemented emission-reduction techniques under ETS, the emission reductions of these firms were assessed based on their emissions relative to the fuel input. As the sites with process emissions were more complex than sites with only combustion emissions, these were assessed separately. Firstly, the reduction of the implied NOx emission factor[5] was calculated on a site level and then by individual installations for the combustion emission sites. The implied emission factor was calculated by dividing the emissions by the fuel input of that year. The fuel input was calculated by dividing the emission rights of that installation by the relevant PSR.

To identify sites for which were most likely to have implemented emission reduction techniques, a 30% site-level reduction threshold was set. For individual installations, the threshold was 40% which concurs with many of the BAT emission reduction techniques in the BREF although omitting some techniques with smaller reduction potentials.

The selection of sites most likely to have implemented emission reduction techniques, resulted in the selection of 83 individual sites. As noted in table 8, 49 sites out of 360 were selected on site level and another 33 sites were selected on the basis of reductions in individual installations. Firms that had participated in the ETS for less than three years were not included in the selection. The total number of sites with only combustion emissions of which the emission reports were individually assessed to find the NOx abatement techniques, was thus 83.

The criterion for sites with process emissions led to the selection of eight out of the in total 27 sites with one or multiple installations which had a reduction of the emission factor of 30% or over. The liberty was taken to also include one site where the emission factor was reduced by 29%. One site was omitted as only one smaller boiler installation was found to have a reduction of the emission factor. The reduction was 77% but as the boiler was only a relatively small 1.7 $MW_{th}$ installation and only emitted 0.3% of the total emissions produced by the site, it was not deemed relevant.

For two sites, emission reductions were identified for the whole site but no data was available on the production of individual units. These sites were, therefore, dropped due to lack of information. Two other sites had sufficient emission reduction on the selectable installations; however, the firms had gone bankrupt and no longer existed. Since no further information could be acquired on the implemented reduction techniques, these sites were not researched further.

Of the sites with combustion emissions which were selected on the basis of emission reductions at site level, the emission reductions of 28 of these sites could be accounted for from explanations provided in the emission reports. Besides these sites, the abatement techniques of another 7 sites which were selected on the basis of reduction on installation level could also be determined from the emission reports. For the remaining sites the contact persons of the company were contacted individually to provide information on the emission reduction techniques.

[5] We refer to implied emission factor due to the emission factor being calculated from activity data, rather than the actual emission factor measured at the installation (UNFCC, n.d.). Unless otherwise stated, when emission factors in this work is referred to, we refer to the implied emission factor. As emission factors are equal to the emissions relative to the fuel input or product output, implied emission factor is occasionally referred to as relative emissions.

**Table 8:** Selection of sites based on either site or installation level and whether the emission reduction was explained in the emission report.

| Level of selection | Number of sites | Reduction explained in emission report? |
|---|---|---|
| Site level | 49 | 28 |
| Installation level | 33 | 7 |
| Process site | 8 | 1 |
| Total | 90 | 36 |

In the following sections the results of the surveys of the emission reports and company survey will be presented. The results of the individual companies have been anonymised as the data on installation level provided by the NEa is confidential, company sensitive data. To anonymize the results, the companies have been codified by industry. Installations within the same site have been noted with an additional code. The codes used for the individual industries and the number of selected sites within each industry is presented in table 9. Industries for which no sites were selected are not included in the table.

**Table 9:** Sector codes per sector, including number of selected sites per sector.

| Sector | Sector code | Number of selected sites |
|---|---|---|
| Petroleum extraction and processing | A | 10 |
| Waste processing | B | 2 |
| Metal production | C | 3 |
| Other | D | 9 |
| Production of electricity, natural gas and heat | E | 26 |
| Production of chemicals | F | 21 |
| Production of pulp and paper | G | 4 |
| Production of food and drinks | H | 11 |
| Production of glass, glass products, and ceramics | I | 4 |

## 4.3 Identification of implemented techniques

### 4.3.1 Techniques deduced from the environmental reports

The analysis of the emission reports showed that many sites were not likely to have retrofit technical measures such as the installation of NOx controlling techniques or by making other technical adjustments to the operating installations. The emission-reduction of 11 sites, including the three sites with the strongest emission reductions, was achieved as a result of the decommissioning of older, more polluting installations and the subsequent commissioning of newer installations. These sites did not then achieve an actual reduction of emissions on the existing installations. These sites included sites E76, D34, E1, E10, E93, H24, H36, E67, G4, E4 and D30.

Several factors other factors were deduced from the annual reports that made it unlikely that new techniques had been installed on several other sites. This concerned sites D78, F4, F6, E16, D72 and G15. For D78 which had a 62% reduction in the site emission factor, the site contained a large number of smaller installations (49 in one year). These were mainly smaller-scale heating boilers most having a lower capacity than 5MWth. It was deemed very unlikely that specific technologies were implemented on these small installations and no account of implemented measures was given in the emission report. Correspondence with the site confirmed that the reduction in emissions were mainly due to the reduced use or decommissioning of installations and increased energy efficiency measures.

Three other sites [F4, F6, E16] were relatively newly commissioned, two having been commissioned in 2011 and one in 2010. All three of these sites had a relatively higher emission factor in the first operating year only. In these cases, it was considered more likely that the NOx emissions were higher in the first year due to a reduced load or as yet not optimized operating conditions instead of the result of retrofitting new techniques in new installations. This assumption is supported by the BREF LCP (European Commission, 2016) which states that new plants are outfitted with primary NOx techniques in their basic designs.

The sites D72 and G15 showed total site emission reductions on a site level but no reductions in the emission factors of individual installations. For these sites it was concluded that it is unlikely that they had installed new techniques. The overall emission-reduction factor in these sites was more likely to have decreased due to the more polluting installations being taken over by cleaner installations for periods of time (for example, reducing the need for a diesel generator).

The drop in implied emission factor for site E71 and B13 was assessed to be due to an increased load of the installation. For site E71, the year in which the emission factor was relatively high, the fuel input was at the least 20 times lower. For site B13, a new installation was put into operation which had a 50% lower input in the first year than in later years; this likely explains the 53% higher emission factor. Finally, two other sites showed to have decreased the emissions of the installation by replacing an older installation with a newer unit (a boiler in G7 and a CHP in D30).

To conclude, 18 sites that had an emission factor decrease of over 30% at site level, were considered unlikely to have implemented emission-reduction techniques. That left 13 sites of which the emission reduction techniques were identified from the environmental report. Only for site F14 and E35 were implemented emission reduction techniques found on an installation level within the environmental report.

**Implemented techniques**

The techniques found in the emission report are provided in table 10 below. While a lot of information could be derived from the emission report of the 15 sites as given in the table, the reports were not always fully clear and complete. Therefore, the emission reduction techniques were also derived based on online sources and expert judgement.

The emission reductions in E13 were derived party derived from the emission report. Within the emission report it was stated that an SCR was placed on one installation. The report stated that an SCR was placed at one installation, however, the other installation showed an equal reduction in the emission factor (80% and 81% respectively). The PLATTS WEP database also showed the second installation was fitted with an SCR. The table was updated with this information accordingly.

A50 is a site with numerous installation, of which 13 installations was shown to have an emission reduction of over 40%. For six installations, the emission report notes the installation of LNB's but all others are not mentioned within the report. A contact at the site confirmed that indeed multiple installations had been retrofitted with LNBs and that there had been a fuel switch. The contact also noted that one installation had been fitted with a secondary DeNOx measure, this likely being an SCR due to the 80% reduction of the installation.

As in the case of site A50, the emission reports of site F14 noted the installation of LNBs on a number of installations, however, several other installations showed similar emission reductions of 55-60%. As these installations were placed within the same production unit of the site, it was assumed that these installations were also outfitted with LNBs.

Site E23 only reported the installation of a low NOx combustion system in a gas turbine installation but did not specify the type of system. As a DLE is considered a BAT and achieves reduction performances within the range of emission reductions seen within E23, it was deemed that this was most likely the abatement option implemented.

For site E96 it has been identified from the environmental report that the instalment of new turbines including a DeNOx was the main source of NOx reduction. The type of DeNOx has however not been identified, both SCR and SNCR are options here. As SCR are regarded as the most advanced technology, this is selected as the option installed. The achieved reduction of the SCR is unclear to, as a new turbine would have lower default emissions even discarding an SCR.

For site E96, the environmental report identified that the instalment of new turbines, including a DeNOx, was the main source of NOx reduction. The type of DeNOx was, however, not identified; both SCR and SNCR are options here. As SCR is regarded as the most advanced technology, this was selected as the option installed. The achieved reduction of the SCR was also unclear since a new turbine would have had lower default emissions even without an SCR.

**Table 10**: Abatement techniques found in the emission report per site.

| Site | Site level Reduction | Technique installed found in the emission reports |
|---|---|---|
| E13 | -78.7% | SCR on two installations |
| E35 | -78.4% | LNB on one installation, decommissioning of one installation and commissioning of another |
| E68 | -78.3% | SCR on one installation |
| E65 | -77.3% | SCR on one and later additional catalyser on SCR |
| A50 | -63.4% | New LNB's on 10 installations, SCR on one installation and a fuel switch to natural gas |
| E23 | -58.4% | Low NOx equipment on three gas turbines |
| D68 | -50.9% | Low NOx burners on 4 boilers |
| H21 | -49% | Two boilers installed with low NOx burners |
| E25 | -42.9% | DLE system installed in GT replacing steam injection |
| E47 | -39.2% | Increased steam injection in a turbine |
| E36 | -33.6% | SCR on one installation |
| E12 | -32.4% | Low NOx equipment on two gas turbines |
| F14 | -4.2% | Flue gas recirculation, improvement of SCR and LNB’s on 8 installations |
| H42 | -28.4 | LNB |
| E96 | -44.1 | 2 new installations, including SCR |

### 4.3.2 Results of the company survey

Of the sites where the emission reductions were not derivable from the emission reports, the individual companies were asked to participate in the survey. 46 sites were contacted, 21 sites were contacted which were selected on the site level, and another 25 on the installation level. Of these companies, 10 companies were unwilling to participate in the research or did not reply. The sites A5, D23, G9, F19, A1, D61, D82, H12, A43 and A47 did not participate, the expected emission reduction techniques will be discussed in section 4.3.3. The results for the process sites will be discussed in section 4.3.4.

Almost half of the surveyed companies (23) noted that although emission reductions were achieved on a site level, installation level or both, that no NOx reducing techniques had been implemented. Some sites were able to provide reasons for the decrease, while others were unable to explain the drop in either the site or installation emission factor. Reasons given by companies included that the decrease was caused by a reduced load of the installation, a different fuel mix (less biomass) or different usage of CHP installations.

One sector which attained significant emission reductions on both site level and installation level, but where no measures were found to have been implemented, was the oil and gas production sector. These sites and the achieved emission reductions are provided in the table 11 below. All sites that responded to the survey indicated that no NOx reducing techniques had been implemented during the time of the NOx emission trading scheme, irrespective of the total reduction in the site emission factor.

**Table 11**: Site-level emission reductions within the off- and onshore industry.

| Site | Emission reduction |
|---|---|
| A33 | -49.3% |
| A40 | -41.2% |
| A34 | -34.5% |
| A36 | -33.3% |
| A41 | -1.1% |
| A31 | 5.1% |
| A30 | -3.2% |
| A38 | 11.1% |
| A39 | 13.1% |

The firms noted that retrofitting offshore turbines with DLE-systems was not cost effective given the low NOx credit price and, as it was not enforced through other legislation, not executed. The firms further explained that the variation in emissions of offshore installations was due to fuel choice and field conditions. When the rigs are in production, the produced gas is used in the turbines and otherwise diesel oil is used. As diesel oil has significantly higher NOx emissions, the switch to gas causes a considerable reduction of the emission factor. Another reason for variability in the NOx emissions is the need for gas compression; if the compression turbines are used at lower loads the emission factors are influenced. Lastly, one company noted the exact emission factor for the installations was not known, so an overestimation of the actual emissions had been made. When the exact key figure was calculated through tests, the actual emissions were shown to be significantly lower.

Several companies in other sectors also noted that their emission reduction was not an actual reduction in emission factor, but due to the calculation method. This was the case for five sites [E34, E21, H22, H4, F9]. Both H48 and H49 noted that the reduction in NOx emission in their boiler was due to a different fuel mix. The installations can be fuelled on both natural gas and animal fat (biomass). Because the addition of animal fat significantly increases emissions, the emission reduction was explained as being the result of a reduced use of these fats. Four other sites noted that the decrease was due to an increase of the load of the installation [F9, H5, HF10, E75]. These reductions were achieved in a CHP and two process furnaces. The decrease in the CHP was stated to be mainly due to the reduced use of the CHP and increased 'import' of electricity. For the process furnaces, the contact person noted that as the units do not have steady loads, the units are provided with two emission factors: one for low loads and one for high loads. The reduction in emission factor was thus due to the use of different emission factors. Three other firms gave no reason for the drop in emission factor, and only stated that no new techniques had been implemented.

Two companies [E2, E33] noted that the achieved drop in emissions came from an improved regulation of the air/fuel input which had led to a respective decrease of 49% and 45% reduction. For these measures, it was concluded that the hardware for the lower NOx levels were already present, but were incorrectly implemented and that this, therefore, is not regarded as an additional measure.

Site E58 had several new measures implemented within the site, achieving an overall emission reduction of 24%. The site with several CCGT units, implemented DLE systems on all CCGTs and installed LNBs on assistance boilers. While information was found within the emission report on the instalment of the DLE systems, the site representative was contacted as the emission reductions of E58-2 did not fit the technique, as can be seen in the below table 12. While the installation is equal to the others, no emission reduction can be identified and the data even shows an increase in emissions. A plausible explanation for this could lie in the installation having decreased the load after the technique was installed, the fuel input for the installations in the table below decreased by over 75%. If the DLE had not been not installed, it is expected that the emission would have been higher, the reduction should, therefore, be calculated using a scenario with a low load where no DLE would have been installed. For this reason, a hypothetical emission reduction of 40% was used on the basis of the reduction at the other installations.

**Table 12:** Emission reductions in the individual installations of site E58.

| Installation | Type of installation | Technique | Reduction |
|---|---|---|---|
| E58-1 | CCGT | DLE | -42% |
| E58-2 | CCGT | DLE | 5% |
| E58-3 | CCGT | DLE | -34% |
| E58-4 | CCGT | DLE | -48% |
| E58-5 | CCGT | DLE | -33% |

**Implemented techniques**

The techniques that were derived from the communications of the site representatives are presented in the table 13 below. In the table it shows that several sites increased their emissions, or did not decrease their emissions fitting with the abatement potential of the technology. This is again due to the site emission factor being an averaged factor including other installations which can have increased their emission factor. For instance, site F32 contained many installations and the 50% reduction of the SCR units did thus not impact the total emissions strongly.

The main implemented techniques were the LNBs on boilers and DLN combustion systems on turbines. The number of implemented techniques presented in the table below are not equal to the number of installations of which a large reduction was shown. The representatives noted only for these installations that techniques had been implemented, noting that no techniques were installed on the other installations.

Site D79 noted that the emissions of a drier were reduced due to the reduced combustion of the flue gas of a boiler. This led to the reduction in NOx emissions of 49%, but it is unclear if this measure was implemented specifically to reduce the NOx emissions or due to other process-oriented reasons.

Site F8 implemented an increase in flue gas recirculation in addition to the ULNB. The 'hardware' for this recirculation was already available and thus the capital costs for this measure would have been relatively limited.

**Table 13**: Implemented abatement techniques found in the company survey.

| Site | Site level reduction | Implemented technique |
|---|---|---|
| D79 | -3.7% | Reduced use of flue gas |
| D81 | 1.7% | Low NOx burner |
| F27 | -47% | Low NOx burner |
| H6 | -46.1% | Dry Low NOx system |
| E55 | 25.2% | DLN system on two turbines |
| F31 | -64% | Dry Low NOx system |
| H6 | -46.1% | Dry Low NOx system |
| F28 | -30.2% | LNB on three boilers |
| F8 | -13.6 | Ultra LNB + flue gas recirculation |

| | | |
|---|---|---|
| E58 | -24% | DLE on 5 installations + LNBs |
| F32 | -9.5% | Two SCR installations |

### 4.3.3 Reduction techniques based on expert judgement

As noted in the previous section, 14 selected sites chose not to participate in the research or did not respond to the requests. For these sites, an expert judgement was made regarding the abatement technique based on the emission reduction, type of installation and consistency over the years of emission factor and fuel input.

Two of the sites which did not respond to the request, were offshore production sites [A1 and A5]. The nine other oil and gas production locations noted that no reduction techniques were implemented and that these techniques were not widely implemented in the sector. Since it is unlikely then that these two locations have implemented measures, these sites were assumed not to have implemented any new techniques.

One other site, D61, had a reduction in the site emission factor of 38%, but only achieved a reduction in an installation of 17%. As a 17% reduction does not fit well with new abatement techniques, it is unlikely that measures other than reducing the use of the more polluting installations were implemented.

For site H12, one CHP installation achieved a reduction in the emission factor of 41%, from around 50-60 grams NOx/GJ to 35 grams NOx/GJ. From the PLATTS WEP database, it can be seen that this installation is already equipped with an SCR installation. However, an online source of the changed emission permit was found online which states that the old installation including SCR was replaced for a new turbine including a DLE system, without SCR. The reduction is thus noted to be due to a new installation and not a retrofit.

For site G9, it was assumed that the emission reduction in one installation (-58%) was due to a change in monitoring technique. This installation was first reported as a cluster of various process-oriented installations. In 2011 the emissions were no longer reported in clusters, but as individual installations. As this falls together with the reduction, it seems likely that the installations have not actually been adjusted but the monitoring has.

Site D82 is a process oriented site, which has several process oriented installations. The emission report describes a relatively complex monitoring plan which stated that the gas usage of the installations was calculated based on an allocation key. The method of calculating the gas usage was changed within the monitoring plan to basing the calculation on historic key figures instead of an allocation key. This change in the way the gas use was calculated likely was the cause in the drop in emission factor, as this change in the monitoring plan was found to be in the same year that the emission of the installation was decreased.  Moreover, the emission factors of the installations were relatively high, being around 150-200 gram NOx/GJ. This high emission level does not correspond with a specific BAT technique. The high relative emission level, combined with the note on the monitoring plan, makes the conclusion that no technique has likely been installed.

Site D23 reached very high emission reductions on both a total site level (-75%) as well as on an installation level (-90% on several installations). The site, however, has only been operational since 2009 and the installations with the highest emission reductions only had a higher emission factor in the first year. With several of the installations only emitting 7-11 gram NOx/GJ it is clear that reduction techniques have been installed, although it is more likely that these were installed at the time of the instalment of the installation rather than retrofitting.

### 4.3.4 Results of the process installations

Of the 27 companies with process PSRs, nine companies had achieved an emission reduction of around or over 30% on one or more installations. As the emission factors for the process installations are calculated per tons product output, a lower threshold was chosen to assess the installations. The selection resulted in the following sites being chosen:

**Table 14**: Emission reductions per selected site with a process PSR.

| Industry | Site | Reduction | Responded to survey? | New techniques? |
|---|---|---|---|---|
| Waste processing | B14 | -71% on production | yes | SNCR |
| Production of glass and glass products | I9 | -32% on production | yes | No |
| Production of chemical products | F47 | -57% on a CHP, -27% on a production installation | No | New gas turbine |
| Production of chemical products | F46 | -38% on one production line | No, bankrupt | No |
| Production of glass and glass products | I3 | -42% on one process installation | No | No |
| Production of glass and glass products | I5 | -29% on one process installation | Yes | No |
| Production of glass and glass products/ ceramics | I7 | -46% on one process installation | Yes | No |
| Production of glass and glass products | I8 | -41% on one production line | Yes | Yes, improved flue gas stream |
| Metals production | C4 | -38% on production line and -57% on an oven | yes | yes |
| Production of chemical products | F40 | 10 combustion installations with -30 to -95% | No | Yes, LNBs |

Due to the more complex nature of these installations and the limited number of firms, all firms were contacted regarding the emission reductions. The emission reductions of B14, however, could be attributed to the installation of an SNCR as indicated by the emission report and this firm, therefore, was not contacted. The subsequent 76% emission reduction is relatively high for an SNCR in the ranges of the BREF LCP, but fits within other SNCR examples (Sorrels, Randall, Fry, et al., 2015)

Four of the five glass producing companies with relatively large emission reduction responded to the survey. Three companies [I9, I5, I8] noted that the emission reductions were not due to the installation of new techniques, but to specific process characteristics. One of the sources of NOx within the glass producing process is the addition of sodium nitrate. This addition oxidizes organic compounds within the raw material for the glass production and reduces the formation of $CO_2$ in the process. This addition, however, acts as source for the forming of NOx within this process and thus increases the emissions. These respondents thus noted that their emissions decreased not due to the installation of any additional equipment, but due to the reduced addition of sodium nitrate. The fourth respondent I7 noted that the emission reduction was due to improved airflow into the flue gas purification plant (SCR) due to a new configuration of the flue gas fan combined with the fixing of leakage in the system. The latter reason is rather specific, as the company noted that that installation was the only installation where the SCR was not placed directly after the glass oven. The 42% emission reduction of I3 falls relatively well within the range of emission reductions in I9, I5 and I8, all of which achieved reductions of 32%, 29% and 41% respectively due to the reduced input of sodium nitrate. It was, therefore, assumed that I3 also achieved this reduction due to reduced sodium nitrate input.

The emission reduction of F46 is not verifiable due to the bankruptcy of the company. However, the emission reports did not report any changes to the installation, but did report the use of several different key figures. As the installation runs on two different fuels, each of which have a different emission factor, the emissions vary per year. This emission factor varies from 0.29 to 1.10 grams/NOx per ton product output. A change in fuel mix can thus explain the changes in NOx emission.

Site F40 is a complex site with multiple different production units. The emission report does not report any changes as the key figures are identified by the individual production units. Significant emission reductions were identified on 11 combustion installations, of which ten installations are steam crackers. From an online source, it was identified that in 2007 only one installation was installed with an LNB while the others were not. The emission factor of the installation installed with an LNB, varied between 25.9 grams/GJ and 33.6 grams/ GJ. The reduction of the other plants led the emission factors of these installations to fall within this range. It was, therefore, assumed that these installations were also outfitted with LNBs. One other 2 MW installation also reached an emission reduction of -57%. As this reduction fits within the reduction rate of LNBs, it is assumed that on this installation too an LNB is installed

For plant C4, the emission reductions are probably both due to the reduction in production and the installation of LNBs. The process cluster had a total emission reduction of 38%. The emission report notes that one whole production facility with over 10 ovens was shut down due to the economic situation. In the same year, the emission factor decreased considerably and it, therefore, seems possible that the shut-down led to the reduction in the emission factor. The installation of LNBs is however plausible for two oven installations. One oven had a 50% higher emission factor than other similar installations in the same year, this was reduced in the following year. A method to achieve this reduction would be LNBs. Moreover, in a permit request found online, it stated that low NOx burners were installed on a smelt oven, reducing the emissions from 195 g/GJ to 100 g/GJ. The reduction found online is higher than the reduction within the dataset, which is only 20%. Assuming the permit has a higher accuracy, this reduction figure is used. A change to the burner is also noted in the emission report. It is thus assumed that the two ovens were installed with LNBs.

The emission reductions in the CHP in F47 were due to the installation of a new gas turbine as stated in the emission report. Without the gas turbine, the CHP was more dependent on a boiler, which led to a higher emission factor. The reduction on the production installation was more likely due to a different emission factor since the emission report notes that the emission factor used leads to an overestimation of the emissions.

## 4.4 Cost estimation of implemented techniques

The outcome of the results of the technique implementation, have been summarized in the table 15 below. The table shows that the main installed techniques are SCRs on large combustion plants (LCP), low NOx burners and dry lox NOx combustion systems on various sized installations, and steam injection and flue gas recirculation on only a few plants. Process furnaces here are defined as installations such as furnaces, ovens and driers. These installations are, however, combustion installations in view of the NOx emission trading scheme since they do not have a process PSR.

As cost information for the installed measures was not part of the company survey, these costs are derived from the calculations presented in the method section. The replacement of old installations by entire new installations has not been integrated in the cost estimations, for instance, in the case of plant F47, a new gas turbine was installed. While this led to lower NOx emissions, the instalment of the turbine is very unlikely to have been incentivised by the NOx emission scheme or the IPPC as the IPPC provides different obligations for older installations. The reduction of NOx emissions due to the switching of fuel, either by using less refinery gas or biomass, has also not been integrated. Installation optimizations, such as improving the air flow into the installation, without implementing any new ‘hardware’ has also not been assessed in the cost estimation. For these cases, the NOx emission may have decreased, but this is not likely to be attributed to the NOx trading scheme or the IPPC regulations. Moreover, the uncertainty of both the costs and abatement effects of these measures make estimations to be of little use.

**Table 15**: Results of the identification of implemented techniques including the average reduction in emissions.

| Type | Average Capacity | DeNOx type | Average reduction | Number of installations |
|---|---|---|---|---|
| Steam turbine | 518 MWe | SCR | -71% | 5 |
| Steam turbine | 420 MWe | Extra catalyser bed | -18% | 1 |
| Steam turbine | 1600 MWth | ULNB | -70% | 1 |
| CCGT | 300 MWe | DLE | -46% | 6 |
| CCGT | 60 MWth | Increased steam injection | -52% | 1 |
| Boiler | 11.14 MWth | ULNB | -63% | 5 |
| Boiler | 17 MWth | LNB | -53% | 9 |
| Cogen | 390 MWth | DLE | -60% | 9 |
| Steam crackers | 44 MWth | LNB | -52% | 24 |
| Process furnaces | 32 MWth | ULNB | -58% | 12 |
| Process furnace | 50 MWth | Flue gas recirculation. | -40% | 1 |
| Process furnace | 21.8 MWth | SNCR | -76% | 1 |
| GT | 80 MWth | Extra catalyser bed | -61% | 1 |
| GT | 6.5 MWth | DLE | -72% | 1 |

### 4.4.1 Cost estimation of SCR and SNCR

Using the cost functions presented in the methods section for SCR, the costs calculated for the coal-fired power plants installed with an SCR are presented in the section below. Both the cost per $MW_e$ are given in the table, combined with the specific costs in euro per ton NOx emission reduced. In the cost calculation, the lifetime of the SCR unit was assumed to be 20 years as used in Vito (2016). The specific costs were calculated both with the capital recovery costs as the annual operation and maintenance costs. Plant E65-1 was initially installed with an SCR which was later updated with an additional catalyst layer. As the cost estimation of this additional layer would be highly uncertain as no cost estimations are found in literature, the cost estimation was made as if the initial SCR had been installed including the additional catalyst layer. SCRs were also installed on industrial units within site A50 and F32. The capital cost of the SCR on the mixed fuel unit was estimated by the formula in section 3.2.4. However, to estimate the O&M cost, data is needed on the electrical output. As the industrial installation do not have an electrical output, the O&M costs are derived from other installations. The share of O&M costs of the capital recovery costs was calculated for the other SCR units and used to calculate the O&M costs for the units in A50 and F32. For the other units, the O&M costs were 42% of annual costs. The factor of 0.42 was thus used to calculate the O&M cost for the SCRs. Due to more strain on the catalysts, the life time of the SCR unit is assumed to be 15 years however, corresponding to the lifetimes stated in the BREF Refinery (EC, 2010).

The cost for the SNCR in site B14 could not be deduced from the formula from Sorrels (2015), the installation has a relatively small capacity which makes the calculation therefor not suitable. To provide a rough estimate of the cost of the SNCR, the average value of 20.03 EUR/kWth of Vito (2016) was used for the investment costs. The O&M costs are also calculated as share of the annual capital recovery costs, as with the SCR in the process installations.

The table shows that the calculated investment costs for SCR units on the coal fired LCP are around 70-75 EUR/$kW_e$ with a more broad range in specific reduction costs, ranging from 2100 to 4800 EUR/ton NOx. The investment costs per MW show slight economies of scale, the larger units have costs around 70 EUR/$kW_e$, whilst slightly smaller units have investments costs around 75 EUR/$kW_e$. The costs per $kW_{th}$ for the industrial units are calculated to be higher however.

The specific NOx abatement costs are mainly dependent on the quantity of NOx abatement. The explanation for the higher costs for E65-1 and E68-1 is that the inlet NOx emissions of these installations are significantly lower than the other three units. The untreated emissions for the first three units is around 120 grams NOx/GJ, whilst the emissions of the latter two units was 71 and 87 grams NOx/GJ. An explanation for this could be existing more advanced primary measures.

**Table 16**: Costs for SCR and SNCR per installation, including costs per kWe or kWth and the specific NOx reduction costs.

| ID nr | MW | EUR/kW | Cost/kW | Spec. NOx reduction cost in EUR/ton NOx |
|---|---|---|---|---|
| E13-1 | 520 | 96.84 /kWe | 1.98/ kWe | 1,978.60 |
| E13-2 | 520 | 97.07 /kWe | 1.93/ kWe | 1,931.57 |
| E36-1 | 600 | 91.78 /kWe | 2.32/ kWe | 2,317.67 |
| E65-1 | 420 | 97.44 /kWe | 3,977/ kWe | 4,388.92 |
| E68-1 | 630 | 91.76 /kWe | 3,629/ kWth | 3,993.62 |
| A50-9 | 126 | 93.89 /kWth | 24,934/ kWth | 25,904.80 |
| B14-1 | 17 | 20.03 /kWth | 624.6/ kWth | 10,901.76 |
| F32-1 | 50 | 130 /kWth | 129.96/ kWth | 33,450 |
| F32-2 | 50 | 130 /kWth | 129.96/ kWth | 33,450 |

It is likely that the above costs per MW show a slight underestimation of the investment costs of the SCR units on the coal fired units. Compared to the cost range of TFTEI (2015), with an average of 122 EUR/MWe and an average of the lower range of 81 and upper range of 139 EUR/MWe, the calculations fit below this range. A manufacturer of an SCR installation however stated that the costs of installingan SCR installation was 25-30 million, putting the cost range towards 40-45 EUR/KWe. It is also unclear if the TFTEI (2015) cost examples only include the investment costs or that the net present value of O&M costs and catalyst replacements are also integrated within these costs. In the above calculations only the investment costs of the SCR unit divided by the unit kW capacity are noted, adding the O&M costs and catalyst replacement costs would show higher cost figures. The specific costs for the industrial unit is difficult to validate as the BREF not only gives specific costs in the range of 2000 EUR/ton NOx, but also costs in the range of 20.000 EUR/ton NOx (European Commission, 2010).

The specific costs for the units not in the power-generating sector are several orders higher than the power producing units. The calculated investment costs of the units within the chemical and refining industries fit with cost estimations of the BREF REF, where the present value of the SCR units in 1998 were around 5-6 million for a 68 MW installation. The total investment costs for the units in F32 was 6.5 million EUR, which fits relatively well. The costs for A50 was estimated to be approximately 11 million, but this installation was double the size of the installation stated in the BREF REF.

## 4.4.2 Cost estimation of LNB and ULNB on boilers and process furnaces

**Boilers**

For industrial boiler units, the only installed techniques which were found were low NOx burners. All low NOx burners which had achieved emission reductions of over 55% were categorized as ultra-Low NOx Burners. The calculations assumed a lifetime of 15 years and zero O&M costs, noting that LNBs give negligible higher O&M costs than conventional burners. The specific costs were, therefore, calculated by dividing the annual capital recovery costs, as calculated in formula 7, by the annual NOx reduction. The cost calculations are shown in the table 17 below. From the table it shows that while the costs per kWth are relatively equal amongst the different units, the specific costs vary strongly.

A slight economy of scale is visible for the costs per MW, though this is slightly obscured due to the additional costs for the instalment of ULNBs. No clear trend is visible for the specific costs since these costs are less dependent on the size of the installation. The specific costs are, however, highly dependent on the level of NOx concentration in untreated NOx emissions. For several installations, the NOx emissions before implementing the (U)LNB were already low at 30 grams NOx/GJ; the implemented measures reduced the emissions still further to attain emission levels as low as around 15-20 grams NOx/GJ. Because the total quantity of NOx reduced with these measures is lower, the resulting specific costs are higher. For these installations, it is likely that a less advanced type LNB was replaced by an ULNB. Higher costs are also achieved if the installation only operates on part-loads. Installations not operating on full-load have lower annual NOx emissions and, therefore, a lower quantity of NOx abatement.

**Table 17**: Costs for LNB and ULNB per boiler installation, including costs per kWe or kWth and the specific NOx reduction costs.

| ID nr | Measure | MWth/ MWe | EUR/kWth or kWe | Spec. NOx reduction cost in EUR/ton NOx |
|---|---|---|---|---|
| E35-1 | ULNB | 100[6] | 2,07 | 484,19 |
| F8-1 | ULNB | 150 | 2,07 | 494,34 |
| A50-7 | ULNB | 51,1 | 2,09 | 204,19 |
| A50-8 | ULNB | 49,6 | 2,09 | 112,51 |
| E58-6 | LNB | 25[7] | 1,76 | 3.716,16 |
| H21-2 | ULNB | 18 | 2,13 | 560,22 |
| H21-1 | ULNB | 14,4 | 2,15 | 203,01 |
| F28-2 | ULNB | 8,8 | 2,20 | 5.073,61 |
| D68-1 | ULNB | 8,4 | 2,21 | 6.247,74 |
| D68-2 | ULNB | 8,1 | 2,21 | 5.497,75 |
| H42-1 | LNB | 7,65 | 1,85 | 499,84 |
| D68-4 | ULNB | 6,8 | 2,24 | 806,71 |
| D68-3 | LNB | 6,6 | 1,87 | 4.681,24 |
| D81-1 | LNB | 4,8 | 1,93 | 4.639,81 |
| F28-1 | LNB | 4,6 | 1,94 | 716,67 |
| F28-3 | ULNB | 2,9 | 2,47 | 1.337,76 |

**Process furnaces**

The cost-estimation of process furnaces have a relatively high uncertainty. The costs of retrofitting the installation make up a large part of the total costs, while the actual burner costs comprise a relatively limited share of the costs. As the retrofit costs of process-oriented installations was found in the literature to be several degrees higher than the costs of LNBs and ULNBs in boiler installations, a different cost function was used based on a range of costs examples as presented in Seebold (2013).

[6] Cost per kWe

[7] Cost per kWe

The results of the cost-estimations of the installation of LNBs in process furnaces are presented in table 18. The table includes five sites, one of which is a refinery, several are chemical firms and one is active in the metal industry. The number of installations to have implemented LNBs is relatively high, as such sites often contain numerous installations.

The cost-estimation in the refinery sector table was used as a proxy for other installations with higher retrofit costs. Such installations are mainly process installations within cracking installations and are thus similar to the petroleum crackers within the refinery sector. However, two ovens and two dryer installations were also included within the *process installation* category. The higher costs for these installations represent the generally higher retrofit costs for this type of installations, but should be treated with caution as they are highly uncertain.

As Seebold (2013) shows linear costs for LNB retrofits in the refinery sector, a linear cost function was chosen based on the cost estimate in the BREF for the Refinery Sector (European Commission, 2015). The investment capital expenses, therefore, do not show any economies of scale. Within the cost functions, again a 15 year lifetime was assumed. The costs per kW thus have a flat rate of 40.86 EUR/kWth, as noted in formula 19, while the specific reduction costs range from 960 EUR/tons NOx to almost 13.000 EUR/tons NOx.

For site F14, a corporate annual report was found that gave the investment costs for the installation of these LNBs to be in the order of 20 million euro's. This cost figure confirms that the cost estimates of at least site F14 are in the right scale, although the estimated costs are below this mark.

**Table 18**: Costs for LNB in process installations, including total capital expenditure and the specific reduction costs per kg NOx.

| ID nr | MWth | Cap ex. in EUR | Specific reduction cost EUR/kg NOx | ID nr | MWth | Cap ex. in EUR | Specific reduction cost EUR/kg NOx |
|---|---|---|---|---|---|---|---|
| A50-1 | 7,1 | 290,127 | 2390,9 | F14-6 | 32 | 1,307,616 | 8144,0 |
| A50-10 | 5,3 | 216,574 | 10531,7 | F14-8 | 59 | 2,410,917 | 13937,1 |
| A50-11 | 8,8 | 359,594 | 6561,6 | F14-9 | 59 | 2,410,917 | 15370,9 |
| A50-12 | 9,3 | 380,026 | 4903,7 | F14-10 | 59 | 2,410,917 | 13446,8 |
| A50-13 | 40,5 | 1,654,952 | 1817,3 | F27-1 | 22 | 898,986 | 6964,9 |
| A50-2 | 23,6 | 964,367 | 4943,3 | F40-1 | 70 | 2,860,410 | 29942,4 |
| A50-3 | 23,6 | 964,367 | 3678,5 | F40-10 | 2 | 81,726 | 9726,2 |
| A50-4 | 8,8 | 359,594 | 8086,1 | F40-2 | 70 | 2,860,410 | 24021,8 |
| A50-5 | 42,5 | 1,736,678 | 1146,2 | F40-3 | 70 | 2,860,410 | 21972,8 |
| A50-6 | 37,7 | 1,540,535 | 2067,5 | F40-4 | 70 | 2,860,410 | 10264,8 |
| C4-1 | 3,5 | 143,021 | 13361,8 | F40-5 | 70 | 2,860,410 | 18668,7 |
| C4-2 | 8,7 | 355,508 | 13238,3 | F40-6 | 70 | 2,860,410 | 12185,1 |
| F14-1 | 50 | 2,043,150 | 5344,5 | F40-7 | 70 | 2,860,410 | 13592,4 |
| F14-2 | 32 | 1,307,616 | 9715,9 | F40-8 | 70 | 2,860,410 | 12681,6 |
| F14-3 | 32 | 1,307,616 | 7295,2 | F40-9 | 70 | 2,860,410 | 20332,2 |
| F14-4 | 32 | 1,307,616 | 8988,9 | F40-9 | 70 | 2,860,410 | 20332,2 |
| F14-5 | 32 | 1,307,616 | 11589,5 | | | | |

## 4.4.3 Cost estimation for DLE and steam injection

In the below table 19, the costs for Dry Low NOx Emission systems and one installation with steam injection (STIJ) are presented for both cogeneration units and specifically CCGT units. Other than for LNBs, the DLE and STIJ specific cost-estimations include the annual O&M costs for the units. The costs per capacity size were calculated based on the electrical generation capacity and the table, therefore, gives the costs per kWe. These costs show significant economies of scale, increasing from 26 EUR/kWe for a 17.5 MW unit to 9.7 EUR/kWe for a 341 WMe unit. The specific NOx reduction costs also reveal a relatively wide cost range being between approximately 600 EUR/ ton NOx and 4000 EUR/ ton NOx. In contrast to the calculations for LNB and SCR, the investment cost estimations were calculated based on the electrical capacity and not the thermal capacity as noted in formula 21. The O&M cost calculation, however, (calculated using formula 22) was based on the full load hours as calculated through the fuel input and the thermal capacity of the installation.

The specific costs are again highly dependent on the relative NOx reduction rate. This rate varies quite considerably between the units: where some units achieve 35% reduction, other units reduce the NOx emission factor by 80%. This discrepancy explains the specific cost difference between E58-5 and E55-1 which are similar types of units but their specific reduction costs largely differ. The achieved NOx reductions are, again, higher for the units with higher initial NOx emissions: the units that achieve reductions of more than 75% all have initial emissions of over 60 grams NOx/GJ, while other units have initial NOx emissions of 20-49 grams NOx/GJ. The average NOx emissions after the measures were installed was 16 grams NOx/GJ.

**Table 19**: Costs for DLE in CCGTs and Cogeneration units per installation, including costs per kWe or kWth and the specific NOx reduction costs.

| ID nr | Type of installation | MWe | Cost/kWe | Spec. NOx reduction cost | Relative NOx reduction % |
|---|---|---|---|---|---|
| E58-1 | CCGT | 341 | 20,20 | 20,20 | -42% |
| E58-2 | CCGT | 341 | 9.69 | 3233,60 | -40% |
| E58-3 | CCGT | 341 | 9.69 | 1986,88 | -35% |
| E58-4 | CCGT | 341 | 9.69 | 1764,39 | -48% |
| E58-5 | CCGT | 341 | 9.69 | 2470,11 | -34% |
| E55-1 | Cogen | 332 | 9.78 | 706,49 | -80% |
| E55-2 | Cogen | 332 | 9.78 | 4719,94 | -41% |
| E23-1 | Cogen | 125 | 13.56 | 1032,41 | -55% |
| E23-2 | Cogen | 125 | 13.56 | 777,17 | -70% |
| E23-3 | Cogen | 125 | 13.56 | 993,49 | -55% |
| E25-1 | CCGT | 78.75 | 15.83 | 1506,45 | -80% |
| F31-1 | Cogen | 59.4 | 17.40 | 1585,60 | -81% |
| E-12-1 | Cogen | 29 | 22.12 | 2220,85 | -45% |
| E12-2 | Cogen | 29 | 22.12 | 2535,05 | -44% |
| H6-1 | Cogen | 17.55 | 26.18 | 1966,98 | -77% |
| E47-1 | CCGT +StIJ | 20.75 | 12.85 | 1923,83 | -52.4 |

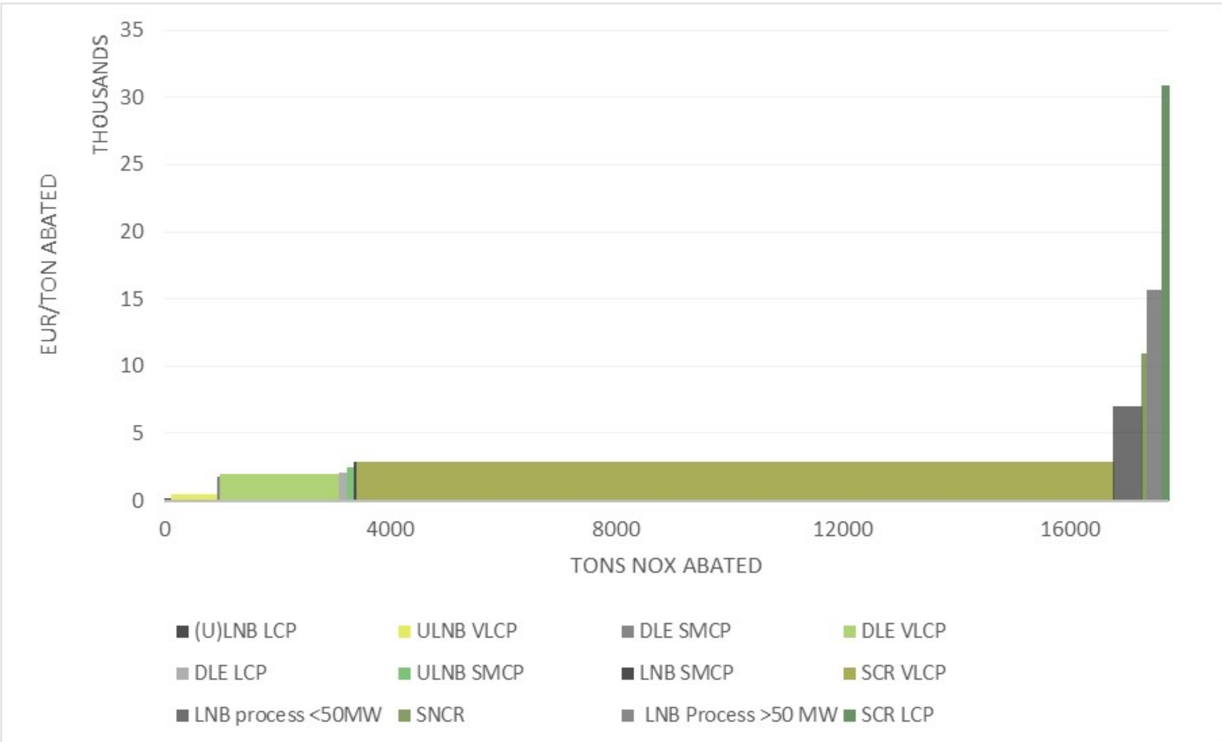
## 4.4.4 Aggregation of abatement measures and costs

The sum of the reduced emissions caused by the various NOx abatement techniques and the consequent marginal, or specific, abatement costs are presented in the marginal abatement cost curve (MACC) in figure 3 below. The various techniques are subdivided according to the different capacity classes, these being small and medium plants (SMCP, <50 MW), large combustion plants (LCP, >50 MW) and very large combustion plants (VLCP, >150 MW). The marginal costs of the category were calculated as the arithmetic mean of the specific costs per ton NOx abated per category.

The width of the bars indicates the total NOx emissions reduced by the measure. This clearly shows the significant impact of the SCRs on coal-fired units but also the relatively high costs of the techniques on small and large process furnaces. Several technologies have had very little impact, the instalment of ULNBs and DLEs on LCPs and the instalment of LNBs, ULNBs DLE on small installations, for example, had relatively low marginal costs but also a relatively low impact on the total abatement. The cost of the SCR on the LCPs had high costs, but low impact; the impact of the DeNOx techniques on the process installations are also relatively limited in the scope of the whole ETS.

The seven installations with the largest NOx reductions as a result of retrofitting abatement technologies, being between 490 and 3547 tons, together reduced the emissions by more than 15.1 ktons of NOx. These installations were all very large combustion plants, five of which were coal fired and the other two fired by natural gas. The techniques implemented to achieve these reductions were SCRs on the coal fired units and DLEs on large gas fired CCGTs. These seven retrofits, therefore, accounted for 83% of the total emission reductions given in the results and for 75% of the reduction in NOx emissions under the entire ETS period. Moreover, virtually all the NOx abatement techniques found were implemented within the power generation and distribution sector which accounted for 93% of all emission reductions shown in the results.

**Graph 3**: marginal abatement cost curve (MACC) of implemented abatement techniques under the NOx ETS.



# 4.5 Comparison of abatement costs

Of the 53.8 ktons of NOx emissions in 2012, 7.8 ktons of NOx was emitted by installations that had implemented NOx abatement techniques between 2006 and 2012. The emissions of the abating companies thus almost exceeded 15% of the total emitted amount. The vast majority of the companies found to have implemented NOx reducing techniques were companies that fell under Annex 1 of the IPPC directive and were thus obliged to implement NOx reducing techniques by the emission limits set by the environmental permits. Only two non-IPPC companies made actual retrofits to their installations, these being sites D68 and D81. Site D79, also a non-IPPC site, did not implement a retrofit but reduced emissions by switching fuel input. The total emissions of the non-IPPC companies that implemented new techniques was 2.2 tons NOx; their contribution to the total reduction of NOx emissions was thus negligible.

Since almost all NOx abatement originated from IPPC companies, it is likely that the main driver behind the implementation of the abatement techniques was the emission limit values. To assess the hypothesis that the interaction of the IPPC put negative pressure on the cost-effectiveness of the NOx ETS, an assessment was made of the remaining abatement potential that was not deployed under the NOx ETS. The assessment was based on the 'least-cost assumption' which was that the cheapest abatement measure would have been the option of choice had the IPPC not enforced emission limit values on the firms. For this scenario, it was assumed that activity levels and, thus, emission rights were equal to the actual historic levels. The scenario, therefore, calculated the cheapest options to reach the achieved emission options.

Due to the high number of installations within the ETS, it was not feasible to make a full analysis of all potential implementable abatement techniques and their subsequent costs. For this reason, the choice was made to centre the analysis on the potentials within the power generation sector. Not only was this sector the largest emitter in the scheme, but it also achieved far greater emission reductions at relatively low cost compared to the process-oriented industries. The power-industry is also the most relevant sector for assessing the interaction of the IPPC on the NOx ETS as it contains the highest number of sites that were not subject to the IPPC, as discussed in section 2.3.

The abatement potential in other sectors is likely to be significant but less cost-effective than in power generation sector. As explained in the introduction to this chapter, alongside the power industry, other main NOx emitting sectors were the refinery, chemical and, metal production industries. As the cost estimations showed that the abatement costs were especially high for the refinery and chemical sectors, it can be anticipated that the costs within the metal industry were not lower: Uylenburg et al. (2012), moreover, showed that the cost estimations of an SCR within TATA steelworks strongly exceeded the estimation of the SCRs on the coal plants.

## 4.5.1 Abatement potential scenario

The installations within the power generation sector which retrofitted abatement measures during the NOx ETS, achieved emission levels under 35 grams NOx/GJ through the use of DLEs, LNBs and SCRs. Only the installation that had implemented steam injection attained a higher emission factor of 67 grams NOx/GJ. The abatement techniques were implemented over 21 individual installations in 11 sites, all of which were covered by the IPPC. The total power sector comprised 76 sites, of which 43 were IPPC sites and 33 sites were non-IPPC sites. The non-IPPC companies were mainly CHP installations that providing district heating.

Although more than 40% of the total number of firms were non-IPPCD, the volume of their emissions in 2012 was only 358 tons NOx. This amounts to only 2.3% of the total sectoral NOx emissions, which is a negligible amount of the total ETS emissions. The meagre contribution of these firms both within the sector as the whole ETS, indicates that their influence on the cost-effectiveness of the ETS would have been minimal. Even if all the non-IPPCD companies had implemented highly cost-effective options, the implementation of abatement techniques in these sites would have been barely visible in the MACC. The expected impact of the IPPCD on the reduction of the cost-effectiveness of the ETS by reducing the implementation of cost-effective technologies in non-IPPCD companies, was, therefore, very low.

However, as the IPPCD only enforces the implementation of abatement technologies reaching emission values at the upper range of the emission limits, it is possible that some firms which otherwise would have implemented more advanced abatement technologies, did not. Moreover, the MACC showed that even though the IPPCD takes cost-effectiveness into account, the implemented measures in process-oriented installations was relatively high.

To assess the abatement potential within the power generation sector, other possible abatement techniques were identified per installation. To assess the feasibility of these additional measures, the techniques already in place were identified based on the emission factor per installation. The following section looks at the abatement potential of coal-fired and gas-fired units. Industrial gas and cokes oven gas power plants are excluded from the discussions as these units are more complex than conventional fuel installations which likely results in higher abatement costs.

## 4.5.2 Abatement potential within the electricity sector

**Abatement potential of coal-fired units**

In 2006, the emission levels of seven coal-fired power plants ranged from 50 to 135 grams NOx/ GJ. The emission levels at the bottom of this range corresponded to installations which had an SCR in place before the NOx Emission Trading Scheme. It is unlikely that additional primary measures could have been implemented in these units at a lower cost than the secondary techniques since had lower-cost primary measures been possible the firms would have undoubtedly opted to implement them rather than an SCR.

The coal plants which, prior to the ETS, were not equipped with an SCR, installed them during the scheme. These newer SCRs were able to attain emission levels as low as 18 grams NOx/GJ. One plant which had already fitted an SCR before the ETS still had emission levels of 64 grams NOx/GJ. This suggests the SCR could have been upgraded with additional catalyst layers to reduce the emissions further. Within the same site, another coal fired unit which was fitted with an SCR reached emission levels of 35 grams NOx/ GJ. If the older SCR had been upgraded to match the emission level of this unit, the potential emission reduction of the abatement measure in 2012 would have been 1 kton NOx. The total abatement potential for the coal fired units is thus the hypothetical 1 ktons together with the achieved 13.3 ktons NOx per year.

The costs of the additional SCR, however, are uncertain. The literature only refers to the costs of the replacement of a single catalyst layer within a SCR unit and not an entire SCR upgrade. If the costs of the upgrade would be equal to the costs of an additional SCR unit, the abatement cost would be in the range of 12.9 EUR/kg, which is significantly higher than the other SCR units. The unit with an SCR already meets BAT and it is not very likely that making major adjustments to the SCR would be very cost-effective before the end of lifetime of the unit.

**Abatement potential for gas turbines**

For the gas turbines, the associated emission levels of implemented DLE systems is between 10 and 20 grams NOx/ GJ. These systems were implemented in installations where initial emission levels were around 30-50 grams NOx/GJ with outliers at 22 and 69 grams/GJ. It was, therefore, assumed that turbines with emission levels above 30 grams NOx/GJ could retrofit DLEs to reduce emission levels. In selecting the installations above 30 grams/ GJ, a number of installations that were found to have implemented abatement techniques while their initial emission was under 30 grams NOx/GJ were omitted from the least cost scenario on the grounds that the achieved total emission reduction was relatively low and the specific costs were high. Installations that were known to be currently decommissioned were also omitted since it is unlikely that a retrofit would be carried out in (soon to be) decommissioned installations.

In 2006, a total of 64 gas turbine installations had an emission factor above 30 grams NOx/GJ. In a frozen emission baseline, these installations would have emitted 6.5 ktons NOx in 2012; but actually emitted 5.8 ktons. The difference between the frozen emission and actual emission indicates that abatement techniques were implemented during the ETS[8].

If the installations with emission factors over 30 grams NOx/GJ had implemented DLE systems and therefore achieved an emission factor at the upper-end of the range (20 grams NOx/GJ), the total of NOx abatement would have amounted to 4.4 ktons NOx. The total abatement potential decreases if only the installations with emission factors over 40 grams/GJ retrofit new DLE systems. This can be seen as a proxy for only retrofitting older installations. The abatement potential in this case decreases to 3.2 ktons NOx.

Although the cost-effectiveness of DLE systems is generally low, the cost depends on the quantity of reduced NOx emissions. Using the cost function as presented in the methods section, the calculated specific costs show several large outliers with high costs. The cost-effectiveness of these systems ranges from below 1 EUR/ tonne to over 100 EUR/tonne for units with very low emissions. The arithmetic mean of all DLE systems is over 100 EUR/tonne.

Assuming that installations with costs over 5 EUR/kg will not implement the measures , the .abatement potential decreases either to 4.1 ktons NOx with an average cost-effectiveness of 2 EUR/kg for all installations >30 grams/GJ or to 3.1 ktons with a cost-effectiveness of 1.8 EUR/kg for those installations with emissions over 40 grams NOx/GJ.

It should be noted, however, that the absolute potential for DLEs is relatively uncertain. Installations with emission levels of around 40 grams NOx/ GJ are likely to have been installed before 1997 and are thus near to reaching end-of-life (Kroon, 2012). It is probable that (a part of) the older installations will be wholly replaced with new installations that include DLE, possibly combined with SCR. The replacement of a fully depreciated installation is, however, highly cost-effective for NOx abatement since the additional costs for implementing DLE in new installations is negligible.

Besides installing DLE as primary technique, secondary techniques like SCRs are also an abatement option for gas turbines. The cost of this option is considerably higher, however, than for primary techniques (Kroon, 2015). As primary techniques are able to reduce the emission levels to 20 grams NOx /GJ, the specific costs of SCRs are considerably higher due to both high investment costs for the SCR and low emission levels of gas turbines. Therefore the installation of secondary techniques on gas turbines are not considered least-cost options.

**Abatement potential for boilers**

The abatement potential of LNBs within the heat and power generation sector is low. Leaving aside the VLCP in E35-1 which was found to have actually implemented ULNBs, emissions from boiler units only amounted to 80 tons NOx in 2012. The power industry mainly operates CCGT installations with DLE as the primary technique, but cogeneration installations with additional firing are also relevant. The NEa database, however, does not distinguish between CCGTs and cogeneration units but only states that a unit is a CHP or gives a unit’s specific name. The actual emissions of burners is thus likely to be higher than 80 tons NOx. The abatement potential, however, remains relatively limited and considerably lower than the potential for SCRs and DLE systems.

### 4.5.3 Comparison of cost effectiveness

As shown in the results sections, abatement costs exceeding 10 EUR /kg NOx were mainly found in process-oriented installations. Under the NOx ETS, 16.7 kton NOx was abated for a price of under 3 EUR/kg while the cost of the abatement of a further 0.9 kton ranged upwards of 7-30 EUR/kg. The total annual abatement costs of these high-

[8] Eight of these installations were indeed found in the result section to have implemented NOx abatement techniques. These eight installations decreased their emissions from the frozen emission baseline from 1.3 ktons NOx to 0.5 ktons NOx. The discrepancy between the reduction of these installations and the total reduction is explained through the relative increase of several installations.

cost measures was 11 million EUR. In a scenario without the IPPC regulations and with the least cost-option, costs above 5 EUR/kg could have been avoided.

The above indicates that for the heat and power generation sector alone sufficient abatement potential was available to reduce emissions further at low cost. If this potential had been fully utilized for the lower end of the DLE abatement potential range, a 3.1 ktons reduction at a cost of 1.8 EUR/kg could have been realised, while at the upper end of the range, the emissions could have been reduced by 4.4 ktons. Accounting for the installed techniques reducing the emissions by 0.8 ktons NOx, the additional low-cost emission abatement potential was 2.3 to 3.6 ktons. If this potential had been fully implemented, the high cost measures could have been 'pushed' out of the market and a total of 19 to 20.3 ktons NOx reduction achieved at abatement costs below 3 EUR/kg. By pushing out only the high-cost options above 5 EUR/kg thereby replacing 0.9 ktons NOx emissions with emissions from the additional DLE retrofit potential, a cost saving of 10 million EUR/year would have been feasible.

# 5

# Discussion

The research question of this study as given below, consisted of two parts: first the effect of policy interactions on the cost-effectiveness of Emission Trading Schemes was to be researched, then the impact of potential interactions on the design of future Emission Trading Schemes was to be evaluated. To assess the results of the study, four hypotheses were posited. The confirmation of these hypotheses based on the results is described in the section 1.1. The impact on future policy design is described in section 1.2.

*What is the effect of policy interactions of the European IPPC directive on the cost-effectiveness of the Dutch NOx emission trading scheme (ETS) and how does this impact future policy designs with similar ETS designs?*

## 5.1 Interaction of IPPC on the NOx emission trading scheme

The results of the identification of installed abatement techniques showed that the techniques installed under the NOx Emission Trading Scheme were almost all implemented in IPPC companies. This confirms the first hypothesis, described in section 2.3, which postulates that mainly IPPC companies would implement abatement measures (see textbox). As only two non-IPPC companies implemented retrofits with a resulting 2.2 tons abated NOx, the influence on abatement levels by companies not subject to IPPC emission limits, was negligible.

The IPPC companies that installed abatement techniques did so even though the price for NOx emissions at the time was very low. It can, therefore, be assumed that the IPPC was the main driver behind the adoption of these techniques.

Moreover, because the emission limit values of the IPPC were more stringent than the prevailing Performance Standard Rate in the NOx ETS, the IPPC companies achieved a greater emissions reduction than would have been attained by limiting the availability of emission rights which was the instrument applied to reduce emissions under the ETS. The effect of the interaction between the IPPC and the NOx ETS was thus an oversupply of emission rights and very low permit prices, which effectively stopped the ETS from functioning. This confirms the second hypothesis.

The specific costs of the abatement measures that were installed in response to the IPPC showed that although the majority of these measures were installed at relatively low cost, a number of more costly measures were installed, too. The total volume of NOx abatement at high cost-effectiveness was, relatively limited, however, being less than one kiloton NOx on a total abatement volume of 18 ktons NOx per year.

**Hypotheses:**

(1) Mainly the companies subject to the IPPC directive will implement measures to reduce their emissions,
(2) The implementation of these measures will put negative pressure on the market price of the ETS,
(3) As these measures will not be the most cost effective, the cost-efficiency of the whole ETS will decrease and,
(4) Since more efficient abatement mixes exist, the potential abatement measures still to be implemented in non-IPPC companies are more cost-effective than measures implemented under the IPPCD.

Within a functioning ETS, the implementation of abatement options with high marginal costs would have been avoided because as the study shows, sufficient abatement potential was available at lower costs. An increased implementation of DLEs in gas fired power plants could have compensated for higher cost-abatement measures in the process-oriented installations. In view of this, it can be concluded that the IPPC obligations caused an increase both of the overall cost-effectiveness of the implemented measures and of the efficiency of the ETS. Hypothesis three is, therefore confirmed.

The remaining low-cost abatement potential was mainly found in IPPC companies; non-IPPC companies had only very limited remaining potential. Because the scope of participants in both the IPPC and the NOx ETS largely overlapped, many participants in the NOx ETS also fell under the IPPC. Those participants not covered by the IPPC had very low total emissions and consequently, their low-cost abatement potential was relatively limited. The fourth hypothesis can, therefore, only partly be confirmed since although low-cost abatement potential was available it was mainly found in IPPC companies even though they were subject to mandated emission limits. The reason for this was that the emission limit values prescribed were at the upper range of the attainable emissions for existing installations, and not the lowest attainable emissions.

In conclusion, the (partial) confirmation of the four hypotheses answers the first part of the research question. The effect of the interaction between the IPPC Directive and the Dutch NOx ETS was a reduction of the cost-effectiveness of the Scheme. This conclusion fits well within the literature on the efficiency of emission trading schemes as posited in the theory section. While it can be concluded that the interaction between the two policies led to a less than optimally efficient distribution of abatement options, whether the benefits of the cost-effectiveness of a sole-ETS scenario outweigh the regulatory costs of the scheme is not shown. This is of importance because even though the regulatory policy puts negative pressure on the cost-effectiveness of the ETS, the regulatory costs may still outweigh this effect.

The marginal abatement cost curve in graph 3 shows that, with the exception of the outliers of the process installations, the abatement costs for NOx abatement techniques are relatively equal and thus have a very low cost-heterogeneity. The number of options for reducing NOx emissions in stationary combustion sources is limited to a handful of primary measures and, basically, only two end-of-pipe measures. For a large share of the emission abatement potential, the cost variations between these options in the various different companies was not very high. The cost estimations showed that the IPPC obligations only enforced a limited quantity of NOx abatement at high costs and the potentials analysis shows that only an estimated cost-saving of 10 million EUR a year could be saved by implementing lower cost options.

The results can thus also be framed differently: the relative flexibility of the *range* of BAT emission limits and the integration of cost-effectiveness in the obligations of the environmental permits of individual companies enabled large emission reductions within the industry to be achieved at largely low cost. It is questionable whether the 10 million EUR/year cost saving of a perfectly functioning ETS outweighs the costs of the regulatory burden of the instrument. In the evaluation study of the NOx emission trading scheme, the regulatory costs alone for participants of the scheme was estimated to be 30 million EUR in the first year of the ETS and 7.6 million EUR in the consecutive years (DHV et al., 2007). These costs only include the monitoring and reporting costs of companies. Added to these private costs, are the public costs of control and regulation and legislative costs. Thus, the calculated cost saving of the sole ETS scenario compared to the actual cost-effectiveness is relatively low compared to the administrative burden of the ETS. It can thus be posited that in terms of the costs and benefits of the ETS policy, the gain in cost-effectiveness of the implemented measures as opposed to a CAC policy might not outweigh the cost of implementing and controlling the policy.

This deviation from the posited hypotheses is mainly caused by (1) the sensitivity of the IPPCD for cost-effectiveness and (2) the stated cost-homogeneity of the various NOx abatement options and (3) the large influence of abatement by (V)LCP. Evidence of the former are the results of the cost-estimations of the measures implemented by the IPPCD; the empirical evidence showed that, overall, the IPPCD was able to enforce the least-cost measures and only surpassed low-cost options in a fraction of the total abatement. The IPPCD differs here from 'traditional' CAC policies as it poses disaggregated associated emission levels per type of installation per subsector per industry. Moreover, these associated emission limits (AEL) based on best available technologies (BAT) are set out in the individual environmental permit. The instrument is thus sensitive to the cost-effectiveness of the implemented measures due to cost-effectiveness being a criteria of determining BAT and the individual assessment per company in the permit process. The added cost-sensitivity in the permit is due to companies having the opportunity to prove that the costs of the AEL would be too high compared to comparable companies (Uylenburg et al., 2012).

Since the IPPCD obligations are sensitive to the specific costs of the abatement options and thus enforce relatively cost-effective abatement options, the cost homogeneity of abatement options decreases the potential gains in the policy cost-effectiveness of the ETS. The cost homogeneity refers to the minor cost difference between many of the abatement options. Although only the potential costs of the heat and power generation sector are thoroughly discussed here, the majority of the abatement measures in the MACC were shown to have costs between 2-3 EUR/kg NOx. The strength of an emission trading scheme is that it obliges the low-cost measures to be implemented first. In the case of NOx abatement, the difference between the costs of the different polluters is relatively low, and thus the cost-savings compared to a CAC policy are relatively low. While higher costs for low-load installations did occur, the share of these installations in the total emissions was relatively limited. Moreover, the high marginal costs for these units can be integrated relatively easily in CAC type policies. For instance, a policy can exempt installations that are in operation for less than a certain number of hours from emission regulations. The absence of strong cost-heterogeneity in the NOx ETS make an alternative theory as

These two reasons suggest that, one, the increase of the policy cost-effectiveness of an ETS relative to CAC policies for NOx emissions in stationary industrial sources is limited due to cost homogeneity and two, the remaining potential efficiency of the ETS was reduced still further as the CAC policy itself was a relatively cost-sensitive CAC policy and achieved high efficiency.

Lastly, the ETS was set-up to include a large proportion of the stationary NOx sources, including many small-sized installations as well as the largest power plants. The volume of NOx abatement of the smaller installations within the scheme was shown to be very low. Only the abatement options implemented in very large scale power plants had significant impact on the surplus of emission rights. Thus, even if the emission trading scheme was not influenced by the IPPCD, the market was likely to have been distorted by the considerable influence of the VLCP. The primary part of the total emission abatement originated from a limited number of installations. Within the ETS, the implementation of only one SCR on a coal-fired power plant reduced the total NOx emission by several kilotons, a quantity incomparable to the abatement potential from LNBs. These measures would thus have blown a large amount of 'air' into the emission trading market and, therefore, again could have led to a surplus of emission rights. If fewer emission rights had been available, the PSR would have been lower, actual price forming would have occurred and the participating firms would have had significant market power. This market power could have led to strategic behaviour on the part of these firms and to less than optimal abatement. The idea behind incorporating many participants is to reduce transaction costs by increasing the liquidity of the market (Algemene Rekenkamer, 2013). In such an instance, the market scope would give several firms very high market power, while the exclusion of the largest actors would possibly improve the market functioning. A better option would, of course, be to scale the ETS to European levels to include more actors and thus reduce the dominance of the largest actors. This was, however, advocated by the Dutch Government in the EU but failed (Uylenburg et al., 2012). The notion of the effect of market power in trading schemes is discussed in the economic literature and should be further researched in this context (Cason, Gangadharan, & Duke, 2003; van Egteren & Weber, 1996); this study has not focussed on the effects this market power could have had within the NOx ETS.

## 5.2 Impact on future policy designs

The above shows that the two policy instruments were counterproductive and, in theory, inherently so. The interaction between the IPPCD and the NOx ETS was used as a case study to show the impact of *direct vertical* interaction between overarching CAC policies and ETS'. Though the theory on ETS' postulates that legitimate grounds exist for imposing additional regulations within an ETS (Sorrell & Sijm, 2003), this was not the case for the IPPCD and the NOx emission trading scheme. The interaction between the IPPCD and the NOx ETS resulted, in effect, in lost function of the ETS due to an increased surplus of emission rights The IPPCD caused the largest emitters to reduce their emissions significantly which directly influenced the abatement within the ETS. In this way, the interaction is far less subtle than, for example, is the case for subsidies and taxation. For instance, the IPPCD also includes provisions on energy efficiency which decreases energy consumption and can thus decrease the demand for carbon credits within the EU Carbon ETS. This effect is, however, is less obvious as it does not directly regulate emissions from the stack. The equivalent of the influence of the discussed interactions of the IPPCD in the context of the EU Carbon ETS would be a provision requiring all coal fired power plants to reduce their emissions by two thirds.

Extrapolating the case and confirming that CAC policies negatively influence the efficiency of an ETS, does not mean that an ETS is the panacea. The limited increase in cost-effectiveness only reveals that, in this case, CAC type policies were not significantly less cost-effective than an emission trading scheme. The study showed that the low cost-heterogeneity s and the influence of majority actors within the ETS, strongly decreased the potential to increase the cost-effectiveness relative to CAC policies. Moreover, CAC policies can reach relatively high cost-effectiveness of abatement measures and discriminate effectively between low and high cost options. This study thus confirms the findings of Kaswan (2011, demonstrating that the cost-effectiveness of CAC policies can be significant and should not be *understated.* Policymakers should thus not always opt for emission trading schemes but rather focus on designing CAC policies which reach maximum cost-effectiveness.

When implementing an ETS, however, attention should be paid both to the potential of interactions between policies to limit the abatement options of actors and cost-heterogeneity. The cost heterogeneity should thus be properly mapped and the market should include various actors with variable abatement costs. As it is expected that the potential for implementing emission trading for air pollutants will be limited by the low-cost heterogeneity, emission trading schemes should be designed to include more actors and where possible, more sectors than just industry, in order to increase the efficiency gains.

Lastly, no evidence was found that it matters whether the emission limiting policy originates from institutions within or above the ETS. The scope of the limiting policy, however, does matter; the IPPC limits were relevant for a very large part of the NOx ETS while national limits in the EU ETS would be relevant for only a small section of the whole ETS. The existence of additional emission limits within an ETS and their scope and stringency should thus always be incorporated in the design of the policy.

This study has not encompassed all aspects of interaction within an ETS but focussed on the impact on the cost-effectiveness of the scheme. While in the next section additional limitations of the study are discussed, several aspects leave topics for further research. One aspect which has not been focussed on in this study is the distributary effects of the pollution itself. An interesting topic for further research would be whether the reduced air emissions of the IPPC increased the spatial distribution of pollutants more effectively than would be the case in a sole-ETS scenario. This is of relevance as the marginal damages of NOx are not uniformly mixed and a concentration of several large emitters will be a deterioration to the local environment.

## 5.3 Limitations

This study has used a bottom-up cost estimation to assess the implementation of abatement options within the NOx ETS. To quantify these results, the NOx abatement options were compared to emission reductions in a frozen emission baseline and the costs of the measures were estimated based on cost figures as found in the literature. Finally, the emission abatement potential at the start of the ETS was assessed based on the potential within the electricity sector. This method and thus the results of the study has several limitations and concern the assessment of implemented measures, the cost estimations and the assessment of abatement potential. There are also limitations with regards to the discussion of the results.

Firstly, it is noted that the assessment of implemented abatement measures is limited by the extent the implemented measures were explained in the emission reports and the quality of the response on the company survey. All sites where either the aggregated implied emission factor (IEF) of the site decreased over 30% or the IEF of an installation decreased by more than 40%, were individually assessed. This selection included sites deemed likely to have implemented reduction techniques, however, it is plausible that some sites that had installed techniques were overlooked. As NOx emissions per installation are variable and dependent on several other factors, it is quite likely that installations which would have been relevant for this study were missed. This corresponds to the many companies which reported an emission factor decreased even though no techniques had been implemented.

For two sites in the refinery sector, it is known that emission reductions were achieved, however, it has not been possible to derive the reduction technique. Neither sites included fuel input data per installation, making an assessment of the IEF per installation impossible nor did they respond to the company survey. Therefore, it was not possible to assess the plausible technique as even the type of installation on which the technique was installed was unknown.

Table 20 below shows the volume of annual emission reductions as explained through the results and the emission reductions per sector based on the activity levels and a frozen emission baseline. As noted in the introductory text on a sectoral level, it was mainly the heat and power generation and refinery industries reached considerable emission reductions. These sectors are also well represented in the results. However, as can be seen from the table, not all emission reductions in the petroleum processing industry have been explained. Moreover, the emission reduction of the refinery sector is not only due to the retrofitting of abatement techniques, but also due to fuel switching. The reduction techniques in the refinery identified in this study, amounted to 655 tons of reduced NOx emissions. Based on the reduction of the whole site emission factor, the remaining 215 ktons of NOx reduction in this site was due to fuel switching. The emission report of this company also stated that the fuel switching from refinery oil and gas to natural gas led to significant NOx reduction.

The quantity of emission reduction explained for the heat and power generation sector is almost equal to the expected quantity of emission reduction. This concurs with the sector being well covered in the results, but also poses the possibility that the effects of some abatement techniques is overstated. As the effect of the abatement technique is calculated by the relative change of the implied emission factor and not the actual emissions in mg per volume of exhaust emissions (nm$^3$), other variability could have been included in the calculation;

For other sectors, the emission reductions based on the frozen emission baseline are relatively limited. Whether the emission reductions were due to emission variability, structural changes in the activity of installations (increased or decreased use of CHP) or abatement retrofits is unknown. Some emission reductions were explained for sectors which increased emissions in a frozen emission baseline, this is explained by other sites within the sector having increased their emissions more than the decrease of others

**Table 17**: Expected emission reductions based on frozen emission baseline and the reductions explained in the results.

| Sector | Expected reduction based on frozen emission | Explained emission reduction |
|---|---|---|
| Petroleum and natural gas extraction | 133.40 | - |
| Petroleum processing | 2,954.52 | 655.1 |
| Waste processing | -279.37 | 74.8 |
| Agri- and horticulture | 9.34 | 4 |
| Asphalt production | -59.03 | 0.2 |
| Production and distribution of electricity, heat and natural gas | 16,993.65 | 16,343 |
| Loading, unloading and handling and storage | -10.61 | 385 |
| Metal production | 2.61 | - |
| Production of chemical products | -51.10 | 53.1 |
| Production of pulp, paper and cardboard | 379.25 | - |
| Production of food and drinks | 48.01 | - |
| Other | 163.00 | - |
| Sum | 20,283.67 | 17,515 |

The estimation of the costs presented in the cost section and in the MACC are inherently uncertain. The costs are based on either cost-calculations or on extrapolations of costs of similar projects presented in the literature. Due to the broad range the various literature sources present, the costs should be seen as an approximation of the actual costs. For several technologies, there was more than a tenfold range in costs between different implemented projects. As discussed in the methods, the costs varied strongly according to the retrofitability of the installation and, for more accurate estimations, should be assessed per installation. The data on the characteristics of the installation was very limited as only either the electrical or thermal capacity was known; in some cases not even the installation type was stated. Therefore, it was not possible to make more sophisticated cost-estimations. An example of improvements to the cost estimations would be to disambiguate the investment costs for the technology in different components with different lifetimes; i.e. the *electromechanical* component of an installation will have a lower lifetime than for instance the ductwork and thus will have to be replaced earlier and have different lifetime costs (Daniels et al., 2006).

The cost-estimates, however, fit within other estimates made within the context of the emission trading scheme, where cost ranges are in the range of 2-3 EUR/kg (Daniels et al., 2006; Uylenburg et al., 2012). However, cost-estimates for SCR in industrial settings are also given values of 40-60 EUR/kg, while some estimates for the refinery industry are also lower than the costs presented.

Lastly, as was discussed, sufficient relatively low-cost abatement potential was shown to be available through the assessment of the potential in the electricity sector. This potential is, again, also inherently uncertain as the assessment was solely based on the implied emission factors of these installations. The NEa database was not suitable for more advanced estimations as very limited data was available on the characteristics of the installations. As much of this information is company sensitive, it is questionable whether a bottom-up analysis would be suitable. Moreover, only the abatement potential of the electricity sector was assessed due to time-considerations. Low cost abatement potential in other sectors will, however, also be available, especially in sectors which use gas turbines and CHPs for heat and power generation. These sectors generally include the horticulture sector, but also, for instance, the chemical sector which uses additional CHPs. However, the total emissions of these sectors are smaller compared to the electricity sector and will have less of an impact on overall abatement potential. Abatement potential in process industries are also present, but as discussed, the cost-effectiveness of these measures are likely to exceed the costs of other industries.

In this discussion, it was noted that the IPPC reduced the cost-effectiveness of the NOx ETS but that this effect was limited due to three factors, these being low cost-heterogeneity, the IPPCD being cost sensitive and the existence of actors with much market power. These conclusions were made on the basis of the results of the cost-analysis, but were not corroborated with more extensive modelling. To make these conclusions more robust, a computable economic model could be used but this was outside the scope of this study.

# 6
# Conclusion

This study showed that as a result of the interaction between the Dutch NOx Emissions Trading Scheme and the IPPC Directive, the cost-effectiveness of the ETS diminished. The emission limit values mandated by the IPPC caused companies to increase the implementation of abatement options above the Performance Standard Rate of the ETS. As a result, although the total volume of high-cost abatement was not large, the IPPC effectively stopped the functioning of the ETS and increased the average abatement costs of the implemented measures. Pursuing emission reductions with Command and Control type policies that limit the choice of abatement options within an ETS, without serving other market-correcting purposes is, therefore, inherently counterproductive and should be avoided to increase the efficiency of emissions trading systems.

Emission trading schemes are however no panacea and in the case of the NOx ETS, several factors other than the policy interactions influenced the successful working of the scheme. In particular, the scope of the NOX ETS hampered its efficiency. If the scope had been expanded to include other sectors and geographical areas, the cost heterogeneity could have been increased and the market power of the largest emitters decreased. Although the impact of market power was not thoroughly studied, it is clear that there was a strong market imbalance. Moreover, emission trading schemes with low cost heterogeneity have little to gain from an emissions permit market compared to CAC policies.

Aside from the relatively small amount of high-cost abatement, because of its flexibility, the IPPC successfully mandated the instalment of largely low-cost abatement options. This reinforces the theory that market-based systems are not the only policy instrument to drive low-cost abatement. Concluding, this case study has shown that Command and Control type policies are able to distinguish instruments by their marginal abatement cost and, when well-designed, the benefits of Command and Control policies can outweigh the benefits of an ETS because of the implementation costs of the latter policy instrument.

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