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Life Side

A LITERATURE-BASED ASSESSMENT OF THE EU ETS

RESEARCH REPORT

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Life

A literature-based assessment of the EU ETS

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Foreword

This report on the EU Emissions Trading System (EU ETS) is a deliverable of the LIFE SIDE project (www.lifesideproject.eu) undertaken by the Florence School of Regulation – Climate between September 2016 and December 2018. The ultimate goal of LIFE SIDE is to support policymakers in the implementation and development of the EU ETS. The report has served the policy dialogue that LIFE SIDE has contributed to fostering (and benefited from in turn) and it has been used in two training courses for policymakers held under the same project.

The report has a few intended features. First, it focuses on four topic areas selected (upon exchange with the Expert Group overseeing the project) in consideration of their relevance in the policy debate. The topic areas are: 1) Free allowance allocation, competitiveness effects and carbon leakage; 2) Interactions with other policies; 3) Low-carbon innovation and investment; and 4) The international dimension. The final chapter of the report is devoted to a broader evaluation of the EU ETS according to the core set of criteria recommended by the European Commission for evaluating EU policy action: relevance, effectiveness, efficiency, coherence, and EU added value. Second, the report draws as much as possible on empirical literature, as opposed to theoretical or ex-ante analyses. Third, the report is accessible to a non-specialist readership. Thus, both information on the relevant legislation and references to economic theory are provided to assist the reader in understanding the results and considerations presented.

Executive summary

The main findings of the report are summarised below. They are organised following the order of the chapters.

Free allowance allocation

Compared to the previous trading periods, the allocation rules adopted for Phase III (2013-2020) represent a quantum leap with respect to both economic efficiency and equity. Auctioning has become the default allocation method for the electricity sector and, in general, free allocation is now limited to about 45% of all allowances. Free allocation itself has significantly improved with respect to economic efficiency thanks to the centralisation of the system, whereby discretion in allocation decisions is minimised, and the application of emission efficiency benchmarks. These alone, it has been estimated, reduced the total quantity of free allowances by about 20% relative to Phase II (2008-2012).

Nevertheless, the literature shows that the rule used for identifying the sectors at risk of carbon leakage, within which installations receive 100% of benchmarked emissions, is too lenient. That is, it results in too many sectors being classified as being at risk of carbon leakage, some of which are in fact most likely not at risk. This result is mainly explained by *a)* an obsolete assumption about future carbon prices and *b)* trade intensity, as an indicator of exposure to international competition, being considered on its own independent of carbon intensity. Moreover, free allocation should be calibrated to better account for firms' ability to pass through the costs of regulation. Indeed, the literature shows that significant levels of cost pass-through also characterise some manufacturing sectors, not only the electricity sector. The EU-wide trade intensity indicator is an imperfect proxy for cost pass-through (in)ability, which heavily depends on product- and location-specific characteristics of the market.

Free allocation in the EU ETS also presents features which can affect the dynamic efficiency of the system (i.e., its long-term cost effectiveness), though very few analyses provide relevant empirical evidence. One study shows that closure provisions, whereby installations exiting the system forfeit their allowance endowments, most likely delayed installation exits in the first two trading periods. With reference to Phase III, a second study shows that allocation adjustments based on activity level thresholds, whereby allowance allocations are reduced after major output reductions, induced greater production. In the case of the cement sector (the only sector analysed), they resulted in lower emission efficiency.

As far as equity is concerned, free allocation has implications for distributional effects between countries, between and within industries, as well as between producers and consumers. The centralisation of the system greatly reduced the potential for distortions in the European market, though free allocation based on EU-wide parameters entails some heterogeneity in effective safeguards across countries. Further distributional effects have been offset through special provisions benefiting lower-income member states, including the redistribution of 12% of all allowances to be auctioned, the continuation of free allocation for the electricity sector and, as of Phase IV (2021-2030), the Modernisation Fund. The question of cost pass-through is relevant to equity too. Windfall profits entail wealth transfers from consumers to producers. Moreover, while free allocation is substantially less generous for the manufacturing sectors off the carbon leakage list, no further differentiation is made among them. Thus, differences in cost pass-through not considered by legislation may determine distributional effects across sectors.

The new or amended free allocation rules defined in the reform for Phase IV can be expected to further improve the efficiency of the allocation system. Carbon leakage risk will be assessed more accurately thanks to: a) carbon intensity and trade intensity considered together through a combined indicator; b) possible use of more disaggregated data, and c) possible consideration of complementary qualitative assessments of abatement potential, market characteristics and profit margins. Moreover, allocations will adjust for output variations exceeding +/-15%.

Competitiveness effects and carbon leakage

The empirical literature on competitiveness effects of the EU ETS and related carbon leakage is wide and growing. Yet, only two studies were found directly testing for carbon leakage; that is, testing for whether emissions increased outside the EU as a consequence of the EU ETS. Within the literature, two main sets of works have been identified. The first considers a range of conventional indicators measuring economic outcomes linked to competitiveness, such as profits, exports, sales, employment and productivity. The second looks at the stock market to infer whether investors believe the EU ETS is beneficial or detrimental to profits. The two bodies of literature complement each other nicely.

By far, the most frequently encountered conclusion is that no evidence was found of negative effects on firms' competitiveness or of carbon leakage. Moderate to very low carbon prices provide the first explanation for this outcome. However, the role of generous free allocation, especially considering firms' ability to pass through opportunity costs of emission allowances, is not less important. A few studies find negative effects, but most of these are either characterised by uncertainty greater than ideal (being statistically different from zero only "at the 10% level") or are accompanied by words of caution due to specific data or methodological issues. At any rate, two studies prompt a warning that deserves attention. For Italy and Germany, they find FDI activity to be higher in the regulated sectors more exposed to international competition and less capital intensive. Positive effects are more frequently found, but they are usually explained by windfall profits rather than by increased productivity. Indeed, it remains unclear whether the EU ETS made firms in any sector or subsector more competitive through innovation or energy efficiency improvements. By contrast, some of the studies analysing stock returns are particularly effective in showing that the combination of free allocation and the pass-through of opportunity costs resulted in profit increases. They do so through comparisons of estimated effects: *a)* between regulated and non-regulated sectors; *b)* between dirtier and cleaner producers within the electricity sector; and *c)* across the first two trading periods and Phase III.

The conclusions that can be drawn from this literature present three orders of limitations. First, most empirical analyses still only refer to Phases I (2005-2007) and II. This delay can be relevant considering all the changes that have taken place with Phase III. Second, estimates of sector-specific effects are relatively few. Sector-specific estimates are particularly valuable because they are better suited to inform policy. Third, with few exceptions, existing estimates of competitiveness effects refer to short-term effects. Yet, the most relevant economic losses that unilateral carbon pricing might cause would unfold over the long term.

Interactions with other policies

The empirical literature analysing the interactions between the EU ETS and other policies is not wide. Within this literature, the subset extending beyond the electricity sector is particularly scant. Each of the empirical studies found focuses on one of the following questions: *a)* How much abatement was due to complementary policies rather than to the EU ETS?; *b)* What was the abatement cost of complementary policies?; and *c)* What was the impact of complementary policies on carbon prices?

Concerning the first research question, the findings of different studies are remarkably consistent with each other. They show that much greater proportions of emission reductions were due to renewable energy (RES) and energy efficiency (EE) policies rather than to the EU ETS. Of course, differences in the amount of abatement vary depending on the magnitude of investments in the given country, and on whether only the electricity sector or all regulated sectors are considered. In any case, effectiveness in reducing emissions does not imply cost effectiveness. Here, the relevant studies agree in showing that abating emissions through RES deployment (specifically, electricity from renewables) was generally expensive and most likely costlier than would have been the case using the EU ETS alone. Yet, RES and EE policies can be considered as serving multiple objectives and, if so, the comparison of their abatement costs with the level of carbon prices is simplistic. Moreover, these policies are intended to address market failures, but the benefits that they provide are difficult to quantify. As to the third research question above, only one study has taken it up. The authors find that variables related to marginal abatement cost theory, including electricity generation from renewables, explain only a small proportion of monthly variations in carbon prices. However, given the relatively high frequency of the data used, the long-term relationships linking carbon prices to fundamentals remain untested in quantitative terms.

The efforts of the EU to implement a mix of instruments that efficiently serves the objectives of climate and energy policies are clearly visible. With reference to the EU ETS, the Market Stability Reserve (MSR) tackles the side effect that other policies and, for that matter any relevant external factor, may have on the dynamic efficiency of the system. By introducing some flexibility into the supply of emission allowances, the MSR is expected to shelter the EU ETS from negative demand shocks and, thus, to prevent overly low carbon prices. Different views exist as to whether the MSR will be effective or whether it is the best possible approach in the first place. For the time being, the MSR’s combination of a rule-based mechanism and the periodic revision of its parameters seems an appropriate solution considering the changing environment external to the EU ETS.

Low-carbon innovation and investment

The empirical literature on the effects of the EU ETS on low-carbon innovation and investment is significant in size but fails to offer a full picture of the state of things. Not a single ex-post analysis exists that comes up to the beginning of Phase III. This is regrettable because major regulatory changes, likely to benefit the system's dynamic efficiency, were introduced with the third trading period: notably, the switch from grandfathering to auctioning in the electricity sector and the application of emission efficiency benchmarks. Moreover, the number of econometric studies is particularly small. While both econometric and non-econometric analyses are needed, the relevance of the results in terms of statistical significance is only clear for the former. The shortage of suitable and accessible databases is the main reason for the scarcity of econometric contributions in this field.

Nevertheless, some conclusions can be drawn based on the literature on Phases I and II:

A) While the EU ETS alone did not stimulate major low-carbon investments, it did stimulate investments typically described as small-scale with short amortization times (e.g., three to five years), resulting in incremental emission reductions.

B) One prominent econometric study finds that the EU ETS brought about a substantial increase in the number of low-carbon patents filed by regulated firms. Patent counts as a measure of innovation output are not exempt from limitations, however.

C) In Europe, the observed surge in the total number of low-carbon patenting was primarily driven by rising energy prices. Similarly, energy prices were much more important determinants of decisions on low-carbon investment than carbon prices.

D) Heterogeneity in the propensity to innovate is significant across sectors and countries. However, evidence is scattered and there is not anything like a comprehensive mapping of low-carbon investments across Europe.

E) Low-carbon innovation efforts have focused on production processes much more than on products.

F) Free allocation appears to have hampered low-carbon investments. The main explanation relates to firms failing to recognise the opportunity cost of using free allowances for compliance. Moreover, an issue that largely concerned the electricity sector in the first two trading periods: new-entrant provisions could affect investment decisions by altering the economic ranking of possible investments in alternative technologies.

G) The EU ETS was generally internalised by firms and it successfully induced organisational innovations.

H) Credible long-term emission reduction targets are important triggers for low-carbon investment because they reduce regulatory uncertainty.

Some of the provisions in the recent reform for Phase IV and the MSR address full on the weaknesses of the EU ETS most relevant to dynamic efficiency. Whether these measures will prove effective is difficult to tell. A well-established view in environmental innovation studies, however, does not seem to find sufficient space in the EU's climate and energy strategy. This view is that support to R&D should be strong and complementary to market-based instruments. There is a compelling economic case for complementing the EU ETS with stronger R&D policies.

The international dimension

The international dimension of an emissions trading system (ETS) pertains to the capacity of its regulation to produce economic or environmental effects overseas, whether through a formal linkage with other climate policies abroad or without. Since its inception, the EU ETS has exerted significant influence on the outside world. Under the Kyoto Protocol's (KP) regime, the recognition for compliance purposes of international emission credits generated by the Clean Development Mechanism (CDM) and the Joint Implementation (JI) was the most important initiative. The results of this experience have been mixed. On the one hand, the linkage to the Kyoto Flexible Mechanisms has extended the carbon price signal to countries and sectors not covered by the EU ETS. On the other, the inflow of international credits negatively affected the EU ETS, both through the dubious nature of the projects underlying some credits and by putting further downward pressure on already low carbon prices. The incorporation of Norway, Iceland and Liechtenstein (EFTA countries, members of the European Economic Area) is the second relevant experience in the early years of the EU ETS.

After Phase I (2005-2007), just as the EU ETS was emerging as a reference for similar systems, international negotiations for an agreement on the post-2020 climate change regime came to a standstill. The UNFCCC COP15 in Copenhagen (2009) marked the end of all hopes for a Kyoto-type regime extended to developing countries, but it was also the start of a process that led to the Paris Agreement (PA) six years later. After COP15, despite the difficulties of the international negotiations, the EU did not give up its ambition to be a leader in the fight against climate change and to strengthen the international carbon market through the EU ETS. It first conceived a project for an OECD-wide carbon market, which was not pursued after the proposal for a national ETS was rejected by the US Senate. Then, bilateral negotiations were engaged with Australia and Switzerland to link the EU ETS with the ETSs of those countries. The first linkage eventually did not happen because the new Australian government was opposed to it. The one with Switzerland is currently awaiting the final vote of the two parliaments. In the meantime, the EU has also contributed to international carbon market cooperation by providing capacity building programmes for setting up and managing domestic ETSs. Looking ahead, the PA's regime is radically different from that of the KP, also in ways that have important implications for the international dimension of the EU ETS. First, as many more are the countries that will need to implement policies for meeting their mitigation commitments, the PA multiplies the number of future opportunities for integration with other ETSs. Second, to achieve the goal of climate stabilisation, the PA rests on international cooperation mechanisms, which to a large extent involves the integration of carbon markets.

So far, the small number of experiences with carbon market integration explains the scarcity of empirical studies that can be considered for assessing the international dimension of the EU ETS in quantitative terms. The only works found relate to the recognition of the CDM and JI emission credits. They mainly look at: the use of international credits by regulated firms; the savings and profits realised by the same; and the relationship between their prices and those of the emission allowances. This literature as well is rather scant, however, also because the use of international credits in Phase III (2013-2020) was strongly restricted. As concerns theoretical studies (not specific to the EU ETS), the wide literature on the linking of carbon markets is being sided by an emerging one focused on integration processes, rather than on their outcomes. Indeed, further learning is needed to facilitate the steps of streamlined carbon integration processes and their governance.

A multi-criteria evaluation of the EU ETS

The relevance criterion for policy evaluation refers to the correspondence between the objective of a policy and the needs of society. No doubt the EU ETS is relevant in this sense. The objective of the EU ETS is "to promote reductions of greenhouse gas emissions in a cost-effective and economically efficient manner". The magnitude of emission reductions ensured by the EU ETS is sizeable given the scope and the long-term trajectory of its cap. This is consistent with the scientific consensus on the necessity of reducing and eventually eliminating anthropogenic GHG emissions. As is generally true for any policy, economic efficiency is a principle expected to inform climate policies. Moreover, uncertainty about the exact entity and timing of the damages from climate change calls for precautionary policy approaches: by predetermining maximum emission levels, the EU ETS abides by this principle.

The effectiveness criterion refers to the success of a policy in meeting its objective. The mission of the EU ETS is to reduce emissions and to do so efficiently. Over the long term, efficiency entails low-carbon innovation and investment. The literature indicates that the EU ETS has contributed to emission reductions. However, there is no exhaustive or even highly consistent evidence on the magnitude of emission reductions attributed to the system. Firstly, econometric estimates of emission abatement are limited to Phases I and II. Secondly, while EU-wide applications tend to find emission reductions in the order of 2%-3% (of business-as-usual emissions), a few country-specific studies find reductions in the order of 15%-25%. Importantly, there is as yet no evidence of carbon leakage. As far as low-carbon innovation and investments are concerned, the EU ETS typically stimulated small-scale investments, resulting in incremental emission reductions. It also caused, as one study shows, a significant increase in the number of low-carbon patents filed by regulated firms.

The efficiency criterion refers to the benefit-cost balance of a policy. In the case of a cap-and-trade system, the theoretical conditions for maximum efficiency are *a)* that the price of allowances match the Marginal Social Cost of Carbon (MSCC) and *b)* that emissions be reduced at minimum cost. Certainly in recent years, carbon prices have been lower than most existing MSCC estimates (to be taken as indicative). This has been mainly due to the impact on allowance demand of the economic crisis and of RES and EE policies, given the rigidity of allowance supply. As concerns minimisation of total abatement costs, three factors are considered: efficiency of allowance allocation, efficiency of the allowance market, and the impact of the system on the economy (competitiveness effects). Efficiency in allocation has clearly improved with Phase III, though, arguably, margin for better targeted free allocation remains. The allowance market has generally performed well, and its efficiency is expected to further improve with allowance scarcity. Importantly, the EU ETS has not significantly affected, so far, the competitiveness of the regulated industries.

The coherence criterion refers to whether a policy is consistent with other policies, notably those having similar objectives, and with the relevant international context. The coherence of the EU ETS is thus evaluated with respect to RES and EE policies in the EU (internal coherence) and to the provisions of the Paris Agreement (external coherence). The question is not so much whether the EU ETS is consistent with the existing climate-energy policy mix (it is) as whether this mix is well balanced. However, there is no obvious answer to this question because quantifying the benefits of RES and EE policies other than climate mitigation is all but simple. Looking ahead, the MSR should improve things by partially insulating the EU ETS from the effects of other policies. As to the Paris Agreement, not only is the EU ETS perfectly consistent with its provisions but it can also serve its successful implementation.

The EU added value criterion refers to whether undertaking a policy at the EU level provides additional value compared to what would result from similar national policies. While no counterfactuals exist, positive EU added value has most likely been attained given the economics of emission trading: the wider the coverage of the system, the more opportunities for cheaper emission abatement, which means lower total costs.

Chapter 1

Free allowance allocation

1. Introduction

In cap-and-trade systems such as the EU ETS, emission allowances can be distributed to regulated installations by free allocation, through auctions, or a mix of these two methods. From an economic perspective, free allocation is the approach used for safeguarding the competitiveness of firms exposed to international competition and, thereby, for preserving the environmental effectiveness of the policy instrument.[1] In other words, the purpose of free allocation is to prevent potential negative effects on the domestic economy to the advantage of countries with less stringent carbon

[1] In cap-and-trade systems older than the EU ETS, notably those in the US for SO_2 and NOx emissions (the Acid Rain Program and the NOx Budget Trading Program), free allocation was grounded on property rights principles. The same seemed true for the EU ETS at its origin, but later the theoretical foundations of free allocation shifted from legal philosophy to economics. Ellerman *et al.* (2010) comment on this evolution: "What is new in the European experience is how quickly a consensus was formed to move from free allocation to auctioning. Indeed, the more general lesson from the European experience would seem to be that, while free allocation may be necessary for the introduction of a limit on carbon, the resulting emission rights and associated value are not granted in perpetuity, as has been done in other contexts. Instead, free allocation is a transitional feature, both expedient and equitable, that will yield to more socially valuable uses of the scarcity rents created by the cap. How this radical change in allocation will work out is the new public policy experiment on which the EU ETS will embark as it prepares for the third compliance period."

regulation. If such negative effects occurred, they would also result in increased emissions overseas, what is commonly referred to as carbon leakage.

Giving allowances away is similar but not equivalent to granting tax exemptions. Since they have market value, using preallocated allowances for compliance carries an opportunity cost, equal to the revenue that would be earned if (the corresponding emissions were avoided and) the allowances were sold. It is therefore legitimate to expect that rational profit-maximising firms will factor such opportunity costs in their decisions – as they would with real costs. Depending on the characteristics of the market, it will be possible, for some firms, to increase profits by passing through their opportunity costs to output prices, even if this meant reducing in some measure the volume of output sold. These extra profits, reaped by passing through opportunity costs (as opposed to real costs) are called windfall profits from cost pass-through.

Windfall profits are highly controversial, as they entail a wealth transfer from consumers to producers difficult to justify. In determining free allocation, the system's regulator – the European Commission in the case of the EU ETS – has thus the challenging task of minimising the risk of carbon leakage while both minimising windfall profits and preserving the price incentive for firms to undertake low-carbon investments.[2] These implications alone demonstrate the centrality of allowance allocation in cap-and-trade systems. By defining the rules for allowance allocation, the regulator can alter the eventual distribution of total abatement costs (i.e., the total cost of meeting the targeted level of emissions) and, relaxing certain assumptions, or in a long-term perspective, it may also alter the magnitude of total abatement costs.[3]

[2] In a cap-and-trade system, carbon prices are determined by the interplay between total supply and total demand for allowances. While the second is given by the sum of operators' individual demands, the first is controlled by the regulator.

[3] According to the independence property, the market equilibrium in a cap-and-trade system is efficient and independent of the initial allowance allocation (Montgomery, 1972). A number of factors, however, can lead to the independence property being violated, including transaction costs, market power and price

In the first two trading periods of the EU ETS – Phase I (2005-2007) and Phase II (2008-2012) – almost all allowances were given away based on installations' historical emissions. A fundamentally different allocation system was subsequently adopted. Since the beginning of Phase III (2013-2020), auctioning has been the default allocation method for the electricity sector. Its progressive extension in other sectors is a stated objective for the future. For the time being, though, selective free allocation applies. This is based on: *a)* the identification of the sectors "at significant risk of carbon leakage"; and *b)* on the application of emission efficiency benchmarks. The recent reform for Phase IV (2020-2030) has not radically changed the rules for free allocation but has introduced some noteworthy modifications.

This chapter offers a literature-based economic analysis of free allocation in the EU ETS. Its main contribution is to offer a comprehensive review of the relevant empirical evidence emerging from two distinct strands of the EU ETS literature. One strand is concerned with the efficiency and distributional implications of the rules for free allocation. Within it, some studies focus on the criteria applied for identifying the sectors at risk of carbon leakage (only relevant to Phase III). Others, meanwhile, focus on the workings of emission efficiency benchmarks (also relevant to Phase III), the unintended effects of allocation adjustments (related to activity level thresholds or to new-entrant and closure provisions), or the possible effect of allowance allocation on emissions or output. The second set of works deals with firms' ability in different sectors to pass through increases in production costs directly and indirectly determined by the EU ETS; that is, increases in carbon costs (whether real costs or opportunity costs) and in electricity costs, respectively.[4]

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uncertainty (Hahn and Stavins, 2011). Free allocation can also affect the dynamic efficiency of the system, that is, its ability to minimise abatement costs over the long term, depending on how entries and exits of new and incumbent installations are regulated (Ellerman, 2008).

[4] Empirical studies focused on competitiveness effects or on carbon leakage can be considered a third relevant strand of literature, as they, indirectly, test the effectiveness of free allocation. However, given

In theory, to preserve firm profitability while avoiding generating windfall profits, the quantity of free allowances granted to a firm, relative to its emissions, should depend inversely on the firm's ability to pass through the costs of regulation (Vollebergh *et al.*, 1997; Bovenberg and Goulder, 2000). In practice, this principle is only partially reflected in the current EU ETS rules for free allocation. Electricity generation has ceased to receive free allowances upon clear evidence of high cost pass-through and related windfall profits. For all other sectors, however, precise estimates of cost pass-through rates (PTRs) are very difficult, if at all feasible, to derive. For this reason, the cost pass-through parameter has not been used for calibrating free allocation in a more restrictive and, hence, better-targeted way.

The chapter is organized as follows. Section 2 recalls the methods of allowance allocation. Section 3 describes the evolution of free allocation in the EU ETS. Section 4 reviews the empirical literature relevant to the efficiency and distributional implications of free allocation in the EU ETS. Section 5 reviews the empirical literature on pass-through of carbon costs in the EU ETS. Section 6 concludes.

2. Methods of allocation

Distributing allowances by auctioning has some important advantages that generally make it economists' preferred allocation method (Hepburn *et al.*, 2006). Notably, auctioning is efficient in the allocative sense, it generates fiscal revenues, and it is transparent. For firms in certain sectors, however, the cost of purchasing the allowances may weigh heavily on their international competitiveness, if so potentially resulting in economic losses domestically and increased emissions overseas, i.e. carbon leakage.

In theory, border tax adjustments (BTAs) are an economically efficient option for solving the issue. Using BTAs involves levying carbon taxes on imports from countries

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the large number of such studies and their technical specificities, they are presented separately in the next chapter of this report.

where carbon prices or equivalent regulation do not exist or are less stringent than domestic ones. Ideally, carbon taxes on imports would reflect the difference between domestic carbon prices and those in the exporting countries. BTAs would establish a level playing field for regulated firms and their competitors in countries not participating in the system and without equivalent policies. Yet, in practice, BTAs are hard to implement both for technical and, perhaps even more, for political reasons.[5] Thus, to date, BTAs have remained a policy hypothesis.

Free allocation is, after auctioning and BTAs, theoretically a “second-best” option for addressing the issue of firms exposed to international competition. As allowances are economic assets, whose value depends on their scarcity, free allocation is a naturally contentious domain. Free allocation can have equity implications relating to distributional effects *a)* between countries (in an international system), *b)* between and within industries, as well as *c)* between producers and consumers. Depending on the specific rules that govern free allocation (and on the price of emission allowances), however, such effects can be more or less significant. In this regard, the evolution of free allocation in the EU ETS (summarised in Section 3) demonstrates how better rules can be defined and implemented over time, improving their economic efficiency while also mitigating distributional concerns.

Focusing on the economic incentives associated with different allocation rules, methods of free allocation fundamentally differ depending on whether the allowances are allocated before emissions are generated (ex-ante), based on historical emissions, or after emissions are generated (ex-post), in proportion to the corresponding output. If based merely on historical emissions, the method is called “historical allocation” or “grandfathering”. Methods of the second type, by contrast, are commonly referred to

[5] BTAs are technically difficult to implement because they require accurate estimates of emission intensities of imported products. In political terms, they are difficult to implement because of the fear that they may compromise international trade relations.

as output-based allocation (OBA). With grandfathering, which is the method that largely prevailed in Phases I and II of the EU ETS, the fewer emissions firms generate the more unused allowances they will retain. Thus, firms have an incentive to reduce emissions by improving emission efficiency (i.e. emissions per unit of output) as well as by reducing output. With OBA firms have, instead, an incentive to reduce emissions only through improved efficiency. One advantage of OBA is that the issue of windfall profits does not arise, as there are no opportunity costs to manage in the first place. On the other hand, simulations indicate that the cost of achieving a given reduction target for total emissions is higher with OBA than with grandfathering. Quirion (2009) provides a detailed illustration of the relative pros and cons of grandfathering and OBA, which, however, generally depend on more specific regulatory aspects.

"Benchmarking" usually indicates a second type of ex-ante allocation alternative to grandfathering. With benchmarking, an emission rate that characterises efficient production (according to, say, the X% most efficient installations in the system or according to best available technology) is used as the basis for allocations, with appropriate adjustments for the past or expected production levels. In addition to incentivising future emission reductions, the distinguishing feature of benchmarking is to reward early movers in reducing emissions. Relative to grandfathering, benchmarking is intended to improve both the economic efficiency of the system and its perceived fairness. Grandfathering may be preferred, nevertheless, especially in the first stages of emissions trading. The reason is that grandfathering avoids high initial costs for the participants, and so it is likely to encounter less opposition. The history of the EU ETS bears this out (Convery and Redmond, 2007; Schmalensee and Stavins, 2017).

3. Free allocation in the EU ETS

3.1 Phase I (2005-2007) and Phase II (2008-2012)[6]

In Phases I and II of the EU ETS, determining the total number of allowances to distribute and their allocation was the responsibility of member states. Governments had to specify their allocation decisions in National Allocation Plans (NAPs), which would enter into force only at the end of a reviewing process directed by the European Commission. The purpose of this process was to ensure that final NAPs would meet the criteria set out in the ETS Directive (European Parliament and Council, 2003).

With regard to allocation, the ETS Directive imposed two fundamental limits. First, member states were allowed to auction up to 5% and 10% of their totals in, respectively, Phase I and Phase II. Accordingly, in the first two trading periods, the vast majority of the allowances were given away for free. In fact, many countries decided not to auction any allowances at all. Second, the ETS Directive excluded the possibility of ex-post adjustments to initial allocations (within a trading period), as those could have incentivised strategic behavior, affecting the allowance market.

In general, within the boundaries set by the ETS Directive, room was left for discretionary choices of member states. This means that allocation, while largely harmonised, inevitably presented some heterogeneity across countries (as documented, among others, by Kettner *et al.*, 2008). The decentralisation of free allocation raised concerns about potential competitiveness distortions in the European market, something that could follow on from differing levels of allocations in the same sector in different countries. Moreover, the ETS Directive did not specify how allocation for new installations or for closing installations had to be regulated or should have been

[6] Ellerman *et al.* (2010) is the reference book for the genesis and the development of the EU ETS in its first five years of operation. The book provides a most comprehensive analysis of the background and the existing evidence concerning free allocation and its effects during Phase I and the first part of Phase II.

regulated. Almost all member states opted for various versions of new-entrant and closure provisions, whereby, respectively, new installations are granted free allowances and closing installations forfeit their future already-allocated ones.[7] The heterogeneity of new-entrant and closure provisions across countries may have had an additional distorting effect on internal competition (Ellerman *et al.*, 2010).

3.2 Phase III (2013-2020)

In 2009, as part of the 2020 Climate and Energy Package, a new Directive (European Parliament and Council, 2009) was agreed which reformed the EU ETS with effect from Phase III. The most salient innovation was the centralisation of the system. Accordingly, the total number of allowances – the cap – is now determined at the EU level and, more importantly, a single set of rules governs their allocation. In Phase III, the cap decreases each year by a linear factor of 1.74% compared to 2010 (the midpoint of 2008-2012)[8], reaching in 2020 a level 21% below 2005 emissions. This trajectory is consistent with the 2020 target for the EU's overall GHG emissions. Above all, the method of allocation changed radically.[9]

While the use of auctioning was very limited in the first two trading periods, the second ETS Directive states that auctioning should be the basic principle for allocation. As of Phase III, auctioning has become the default allocation method for installations generating electricity (i.e., their operators must buy all the allowances they need for

[7] New-entrant and closure provisions became a characterising feature of the EU ETS. In previous cap-and-trade systems, with few exceptions, new entrants had to purchase whatever allowances they needed, and the owners of closing installations were allowed to keep their allowance endowments (Ellerman and Buchner, 2007). To the best of our knowledge, Sweden is the only country not to have included closure provisions in the first two trading periods. A compendium of the different new-entrant and closure provisions adopted in Phase I is found in Schleich and Betz (2005).

[8] In absolute terms, the 1.74% reduction factor corresponds to about 38 million fewer allowances being issued each year.

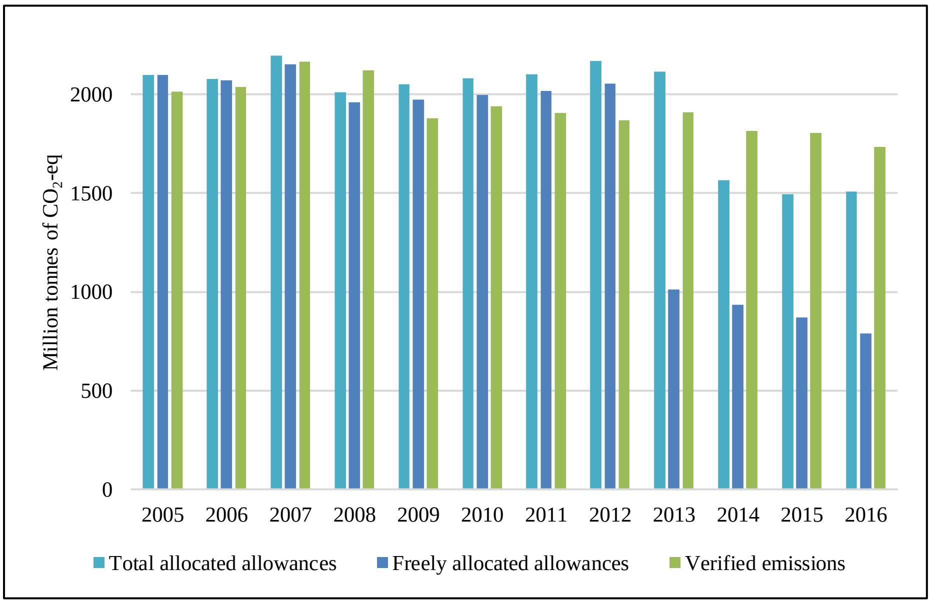
[9] The formula of free allocation in Phase III is reported in the Appendix.

compliance). By derogation, some member states, among those with a GDP per capita below 50% of the EU average, can continue free allocation to installations generating electricity in exchange for their modernisation of the sector and for the diversification of the energy mix. [10] For all other installations, benchmarked allocations were introduced (European Commission, 2011).

The application of emission efficiency benchmarks (expressed in terms of emissions per unit of output) is combined with the identification of the sectors at risk of carbon leakage. Installations in the sectors "deemed at significant risk of carbon leakage" are given free allowances covering 100% of their efficient level of emissions (or "100% up to the benchmark"). The efficient level of emissions is determined by multiplying the relevant benchmark by the installation's recent output level. In Phase III this is the highest level between median production over 2005-2008 and median production over 2009-2010. As a rule, the benchmark corresponds to the average performance of the 10% most efficient installations over 2007-2008. The European Commission developed 54 benchmarks (52 product-specific and two fallback approaches based on heat and fuel consumption). For all other installations, that is, those not falling in the sectors at risk of carbon leakage, free allocation is significantly less conservative. Between 2013 and 2020, free allowances cover progressively smaller shares of installations' efficient level of emissions (defined as above), namely from 80% to as little as 30%.

[10] In addition, the 2009 ETS Directive introduces a redistribution mechanism operating through cross-country allocation of the allowances to be auctioned. 88% of these – the bulk – are allocated to member states in proportion to their 2005 verified emissions. 10% are distributed to the least wealthy member states as an additional source of revenue to help them invest in climate change mitigation and adaptation. The remaining 2% are given as a bonus to member states which by 2005 had reduced their emissions by at least 20%, compared to their base year under the Kyoto Protocol.

Figure 1 - Total allowances, free allowances and emissions[11].



Source: European Union Transaction Log.

The identification of the sectors at risk of carbon leakage is based on two sectoral indicators which are computed at the aggregate EU level: Carbon (cost) Intensity (CI) and Trade Intensity (TI). CI measures the potential significance of carbon costs as in the maximum impact that carbon prices could have on the sector. It is defined as the ratio between *a)* the sum of direct and indirect (i.e. generated through electricity consumption) emissions valued at €30/tCO_2 and *b)* the gross value added. TI, which measures the openness of the sector to international competition, is a proxy for the inability to pass through additional costs without loss in international market share. It is defined as the ratio between *a)* the total value of exports and imports (respectively, to and from non-EU countries) and *b)* the total EU market size, which is equal to the sum of turnover and imports (European Commission, 2009). A sector is then classified

[11] In this graph, the drop in "Total allocated allowances" observed in 2014 reflects the postponement of the auctioning of 900 million allowances. This short-term measure for dealing with the excess supply of emission allowances, known as "backloading", is discussed in Chapter 3.

as being at risk of carbon leakage if one or more of the following thresholds is exceeded: CI >30%, TI >30%, and the double threshold CI >5% & TI >10%. The first carbon leakage list was defined in 2009, for the years 2013 and 2014. Out of 258 sectors, 165 were classified as being at risk. [12] The second carbon leakage list was defined in 2014 for the years 2015-2019.

Because the aggregate amount of preliminary free allocation calculated by member states exceeded the maximum amount of allocation available, a uniform cross-sectoral correction factor (CSCF) is applied to all installations. In 2013, the CSCF reduced the total number of free allowances by 5.7% (European Commission, 2013). As the cap declines over time, the CSCF increases with it, reaching 17.6% in 2020.[13] The application of the CSCF implies that even the most efficient installations do not receive enough free allowances to cover 100% of their initial emissions[14].

3.3 Phase IV (2021-2030)

In March 2018, the reform for Phase IV became law after over two years of negotiations (European Parliament and Council, 2018). The reform has three main objectives: strengthening the price signal by reducing the cap at a faster pace[15,16], better-targeting free allocation, and supporting low-carbon innovation and energy sector

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[12] The list was amended a few times (in 2011, 2012 and 2013) to include more sectors.

[13] As a result of the judgment of the European Court of Justice of 28 April 2016, in January 2017, the European Commission recalculated the CSCF factors. The new values are higher than the previous ones, but they are applied only to decisions on free allocations adopted after 1 March 2017. Hence, for the vast majority of installations, the relevant values will remain the initial ones.

[14] The formula of free allocation in Phase III is reported in the Appendix.

[15] The linear reduction factor applied in Phase IV will be 2.2%. The cap reaches in 2030 a level 43% below 2005 emissions. This level is consistent with the EU's mitigation target for 2030 (40% reduction of overall GHG emissions below 1900 levels) set in the 2030 Climate and Energy Framework.

[16] The strengthening of the price signal is also pursued through the enhancement of the Market Stability Reserve mechanism. For an analysis of how changes in different key parameters of allowance supply would impact on allowance price paths in Phase IV and beyond, see Perino and Willner (2017).

modernisation (in lower-income member states) through dedicated funding mechanisms based on auction revenues (the Innovation Fund and the Modernisation Fund).

Focusing on free allocation, the main new provisions are the following. First, as a rule, a sector will be classified as being at risk of carbon leakage if the product of the carbon *emissions* intensity indicator (CeI) (expressed in terms of $KgCO_2$ per Euro of gross value added) and the TI indicator, $CeI \times TI$, exceeds 0.2. As this rule is more stringent than the one applied in Phase III, the number of sectors classified as being at risk will shrink[17]. Second, to better align allocations with actual production levels, allocations will be adjusted in case of output variations exceeding +15% (increases) or -15% (decreases)[18]. Third, to limit the use of the CSCF (if needed), the share of allowances to be auctioned will be reduced instead of applying the CSCF, by up to 3% of the total quantity of allowances.

4. Efficiency and distributional implications of free allocation

This section reviews the empirical literature providing evidence relevant to the economic efficiency and distributional implications of free allocation in the EU ETS. The works surveyed (12) are divided into two groups and presented accordingly. The first set of studies focus on the criteria that, as of Phase III, have been used for identifying the sectors at risk of carbon leakage. The second set of studies is diverse in terms of the topics addressed. It includes analyses of the benchmarking system (also in force as of Phase III), of the distortionary effects of activity level thresholds, as well as

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[17] According to the European Commission, of the 177 sectors currently classified as at risk, only about 50 will continue to be classified as such. However, the reduction in terms of emissions is smaller than the reduction in terms of sectors (European Commission, 2015b).

[18] Variations are relative to production levels considered for determining initial allocations. They are calculated as rolling two-year averages.

tests of the hypothesis – central in the economics of cap-and-trade – that firms' operational decisions do not depend on the allocation method.

4.1 Identification of the sectors at risk of carbon leakage

As of Phase III, the success of free allocation in safeguarding firms' international competitiveness critically depends on the criteria applied for identifying the sectors at risk of carbon leakage. Four empirical studies were found assessing such criteria based on the resulting selection of the sectors deemed to be at risk. Using different approaches, the studies are concordant in concluding that the current identification method is too lenient. That is, it results in too many sectors being classified as at risk of carbon leakage, some of which are in fact most likely not at risk.

The study by de Bruyn *et al.* (2013) is chronologically the first of the four. The first official list of sectors deemed at significant risk of carbon leakage was compiled in 2009. It comprised 60% of all regulated sectors, representing about 95% of industrial emissions. Using the same data and methodology of the European Commission, de Bruyn *et al.* (2013) find that the number of sectors classified as being at risk would shrink by 50% if the calculations were updated to the new economic context. In terms of emissions, the difference would be even bigger, as free allocation would regard only 10% of industrial emissions. Specifically, two parameters are revised. First, the authors assume average carbon prices in 2020 to be €12/tCO_2, instead of €30/tCO_2. Second, they assume the benchmarks to be less stringent (given technological progress), leading to only 20% of industrial emissions being above the benchmarks, instead of 60%. In addition, the sectoral indicators of trade intensity are recalculated excluding the trade flows between the EU and those countries that, in the meantime, have joined the EU ETS, namely Norway, Liechtenstein, Croatia and Iceland.

Using the responses from interviews with more than 400 managers of regulated firms in six European countries (Belgium, France, Germany, Hungary, Poland and the

UK),[19] Martin *et al.* (2014a) measure individual firm vulnerability to carbon leakage. On a five-point scale, the vulnerability indicator measures managers' expected risk of closing or relocating the firm outside the EU because of carbon pricing. The results indicate this risk to be generally low.[20] The study makes the important point that free allocation is efficient if it equalises, across firms, the marginal propensity to relocate weighted by the damage from their relocation. Depending on the government's objective, this could be the displacement of emissions or jobs, or a combination of the two, overseas. The problem is that firms' propensity to relocate is not publicly observable and, if self-stated, it could be manipulated. Yet, the authors demonstrate through simulations that simple allocation rules based on observable firm-level variables, such as emissions and the number of employees, can be more efficient than the allocation system in force.

Drawing on the same interviews of Martin *et al.* (2014a), Martin *et al.* (2014b) explore the correlations between the indicator of vulnerability to carbon leakage (or relocation risk or propensity) used in the companion study and those of trade intensity (TI) and carbon intensity (CI) used by the European Commission. Self-stated relocation risks are found to be strongly correlated with CI but not with TI. However, most sectors are classified as being at risk exclusively because TI is high. Two amendments to the currently applied criteria are thus proposed. First, sectors with high TI should qualify as being at risk only if CI is high too. Second, in the calculation of TI, only trade with less developed countries (rather than with all non-EU countries) should be considered. If these criteria were applied, the number of sectors classified as at risk would halve.

[19] The interviews were carried out in the second half of 2009.

[20] The sectors that are the most vulnerable are: other minerals, glass, iron and steel, and cement, the respective mean values of the vulnerability indicator ranging between 2.5 and 3.5. Other sectors, such as food and tobacco, fabricated metals, and vehicles, are significantly less vulnerable than the average.

Sato *et al.* (2015) also analyse the robustness and, in general, the aptness of the CI and TI indicators used by the European Commission. Using national statistics for Germany and the UK, the authors show the significance of variation both in sectoral CI and TI across countries and, therefore, the inefficiency of EU-aggregate CI and TI indicators in identifying sectors at risk at the national level. Cross-country variation in CI is due to differences in production processes, technologies and fuel mix,[21] as well as in sector classifications and data quality. Sectoral TIs are also shown to vary considerably across countries. Moreover, TI is criticised for being only a very imperfect proxy for cost pass-through inability. The authors recommend that policies addressing carbon leakage should be more focused, targeting narrower subsectors. While such policies would remain homogenous in the EU, more factors should be considered for assessing carbon leakage risk at a less aggregate level.

4.2 Further topics: benchmarks, allocation adjustments (closure provisions and activity level thresholds), endowment effects (independence property)

Sartor *et al.* (2015) investigate the distributional and economic efficiency implications of free allocation in Phase III. Three findings are of special interest. First, it is estimated that the application of benchmarks alone reduced the total quantity of free allowances by about 20% (between 13% and 24% across sectors) relative to Phase II, while still sufficiently shielding energy-intensive industries from the risk of carbon leakage. Second, following the switch to the Phase III allocation regime, the redistribution of free allowances across sectors is found to have been small between countries and much more significant within countries. Third, redistribution within countries brings with it important benefits. Indeed, the authors show that, in Phase II, a significant degree of heterogeneity in the allocations granted to installations producing cement was not

[21] The authors also indicate more specific factors, including, e.g., cross-country sectoral differences in process emissions, recycling rate and product mix.

explained by historical emissions. Overall, these results indicate that the EU ETS clearly benefited from the introduction of emission efficiency benchmarks and the harmonisation of allocation rules both with respect to economic efficiency and equity.

Stenqvist and Åhman (2016) focus on the performance of emission efficiency benchmarks, showing how well-suited these can be for homogeneous sectors, but significantly less so for more heterogeneous ones. The allocation outcomes of applied benchmarks are analysed and contrasted for the cement industry and for the pulp and paper industry in three countries, namely Sweden, France and the UK. While in Europe the cement sector is highly homogeneous, the pulp and paper sector is highly heterogeneous in products, infrastructures and fuel mixes.[22] In this sector, the use of benchmark values biased towards a fossil fuel-mix and based on energy use rather than on emission intensity leads to allocations that do not necessarily represent the average performance of the top 10% of emission-efficient installations. Sweden's pulp and paper sector provides a striking example, as said imperfections of the benchmarking system result in significantly higher allocations than in Phase II.

Verde *et al.* (2018) show that installation exits from the EU ETS (due to closure or partial closure) were concentrated in the final years of Phases I and II (2007 and 2012). Since closure provisions create an incentive to delay exit, the authors investigate whether indeed those explain at least in part the observed pattern of installation exits. A hazard model for the risk of exit is estimated, controlling for several installation and firm-level variables, as well as for higher-level factors. On average, estimated risk evolves differently depending on whether an installation receives free allowances. Strong evidence of the delay effect is found, as only for installations receiving free allowances is the risk of exit time-dependent, steeply increasing the final year of a trading period, especially Phase II.

[22] Out of 52 product benchmarks, eleven are specific to the pulp and paper sector, which only makes 2% of EU ETS emissions.

To reduce windfall profits from overallocation, as of Phase III, installations whose output falls below 50%, 25%, or 10% of the historical level (used for determining the ex-ante allocation) see their allocation reduced by, respectively, 50%, 75%, and 100%. However, when confronted with low demand, producers may respond to these thresholds by adjusting output strategically. Comparing emissions in the cement sector before and after activity level thresholds were introduced, Neuhoff *et al.* (2014) point to this type of strategic behaviour. Their interviews with managers confirm their hypothesis. Branger *et al.* (2015) extend the analysis and estimate that, in 2012, the activity level thresholds (ALTs) induced excess cement clinker production of 6.4 million tonnes (about 5% of total EU output). This resulted in distorted trade patterns and, in some cases (notably Spain and Greece), reversals of carbon intensity improvements. As intended, ALTs reduced free allocation by 4%, but – the authors observe – output-based allocation would have achieved a 32% reduction.

Finally, in the cap-and-trade context, the endowment effect refers to a situation in which free allocations affect production decisions thus constituting a violation of the independence property (Hahn and Stavins, 2011). The endowment effect is usually explained with firms undervaluing (not fully internalizing) the opportunity cost of using free allowances for compliance. Its presence indicates a loss in cost-effectiveness. Reguant and Ellerman (2008) exploit a non-linearity in the allocation rule for coal plants in Spain, in Phase I, to test for the relevance of initial allocations to emission abatement outcomes. The evidence suggests no systematic relationship between the initial endowment and production decisions. Zaklan (2016) investigates the same question. Exploiting time and cross-country differences in allocation rules, he tests whether, with Phase III, the switch from free allocation to auctioning affected emissions from electricity-producing installations. No evidence of an endowment effect is found save for a subsample of small emitters. The author also argues that endowment effects are more likely to occur for manufacturing installations, especially small ones, as they tend to be less active traders than electricity producers. This conjecture finds support in the analysis by De Vivo and Marin (2018), who test for whether granting more generous allocations to the installations in sectors classified as being at risk of carbon leakage affected their emission abatement behaviour. It turns out that indeed these installations appear to have reduced emissions less than other regulated manufacturing installations. The authors also suggest, however, that exposure to international competition in manufacturing sectors means that some firms may prefer not to fully pass through opportunity costs.

5. Cost pass-through

The impact of the EU ETS on regulated firms depends critically on the ability of these to pass through the costs of regulation. The usual measure of cost pass-through ability is the Pass-Through Rate (PTR), which quantifies the change in output prices relative to a given cost shock. The PTR is expressed in percentage terms, so that, for example, in the literature that interests us, a carbon cost PTR of 85% indicates that a €1 increase in carbon prices results in a €0.85 increase in output prices.

Our literature review identified 20 econometric studies providing estimates of cost PTRs for different sectors covered by the EU ETS. Most of these studies (14) deal exclusively with the electricity sector and elaborate PTR estimates using data from Phases I or II or both (Table 1). The prevalence of studies for the electricity sector is explained by data availability and quality as well as by the expectation of high PTRs vis-à-vis free allocation in the first two trading periods.

Table 1 – Time data coverage of the reviewed literature on cost pass-through.

Study	Phase I			Phase II					Phase III			
	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Electricity sector												
Sijm *et al*. (2006)												
Honkatukia *et al.* (2006)												
Fezzi and Bunn (2009)												
Sijm *et al.* (2008)												
Fabra and Reguant (2014)												
Zachmann and Hirschhausen (2008)												
Chernyavs'ka and Gullì (2008)												
Fell (2010)												
Thoenes (2014)												
Jouvet and Solier (2013)												
Freitas and da Silva (2013)												
Hintermann (2016)												
Huisman and Kilic (2015)												
Freitas and da Silva (2015)												
Manufacturing sectors												
Alexeeva-Talebi (2011)												
Oberndorfer *et al.* (2010)												
Alexeeva-Talebi (2010)												
De Bruyn *et al.* (2010a,b)												
De Bruyn *et al.* (2015)												

Drawing on the same set of studies, Table 2 shows the distribution of estimated PTRs by country and sector.[23] Germany is the country for which PTR estimates are most numerous. Across countries, the electricity sector is the most frequently analysed (particularly) in terms of the number of PTR estimates (26). The refining sector, which comes in second, presents the widest country coverage, though largely thanks to a single study (by Alexeeva-Talebi, 2011).

[23] Some studies provide estimates for more than one country or sector.

Table 2 – Number of PTR estimates in the reviewed literature.

Country	Electricity	Cement	Ceramics	Chemicals	Glass	Iron and steel	Pulp and paper	Refining	Total
DE	6	2		1	1		1	2	13
UK	3	1	1		1			1	6
FR	1	1			1			2	5
IT	2				1			2	5
ES	3				1			1	5
NL	2							1	3
SE	2							1	3
CZ	1	1						1	3
PL	1	1						1	3
BE								2	2
FI	2								2
DK	1							1	2
PT	1							1	2
GR								2	2
AT								1	1
HR								1	1
LT								1	1
NO	1								1
EU	1			3		2			6

Note: Some studies estimate more than one PTR for more than one sector/country.

In the following, the results of this literature are illustrated. The first half of the review is devoted to the studies offering PTR estimates for the electricity sector. The second half covers the studies relevant to the manufacturing sectors.

5.1 Electricity sector

In terms of emissions, the electricity sector is the largest of all those under the EU ETS and the largest, more generally, in the economy. The electricity sector is central in the production system and, of course, electricity is a basic consumption good. What is more, because of its market characteristics, a high pass-through of carbon costs is expected in this sector. For these reasons, estimating the extent to which carbon prices are effectively passed through to electricity prices is critical for the wider assessment of the EU ETS.

The empirical literature confirms that, in Phases I and II, the opportunity costs of free allowances granted to electricity producers were largely passed through to output prices. Two features of electricity markets explain why this could happen: the very limited exposure of producers to international competition and the low elasticity of electricity demand. An important implication is that electricity producers were able to profit from regulation.[24] However, as we, by now, know well, apart from transitional free allocation accorded to lower-income member states, free allocation to the electricity sector ceased with Phase III.

Table 3 – Estimates of carbon cost PTRs for the electricity sector.

Study	Period	Country	Electricity price (spot/forward)	Approach	Average PTR	Peak PTR	Off-peak PTR
Sijm *et al.* (2006)	2005	DE, NE	F	SEM	-	117% (DE), 78%(NE)	60% (DE), 80%(NE)
Honkatukia *et al.* (2006)	2005-2006	FI	S	VECM	75-95%	-	-
Fezzi and Bunn (2009)	2005-2006	UK	S	VECM	32%	-	-
Sijm *et al.* (2008)	2005-2006	FR, DE, IT, PO, ES,SW, CZ, NE, UK	S and F	SEM	38-134% (F), 0-200%(S)	-	-
Fabra and Reguant (2014)	2005-2006	ES	S	SEM	86%	100%	60%

[24] The literature identifies four types of EU ETS-induced windfall profits: *a)* from pass through of opportunity costs; *b)* from over-allocation; *c)* from exploiting the price differential between emission offsets issued under the Kyoto Protocol's Flexible Mechanisms and emission allowances issued under the EU ETS (de Bruyn *et al.*, 2016); and *d)* from inframarginal rents, arising when marginal producers setting output prices use more carbon-intensive technologies. For the electricity sector, windfall profits are mainly reaped through cost pass-through and inframarginal rents. Keppler and Cruciani (2010) estimate that, in Phase I, €19 billion windfall profits per year accrued to the European electricity sector, of which €13 billion were from the pass-through of opportunity costs.

Zachmann and Hirschhausen (2008)	2005-2006	DE	F	VECM	Asymmetric PTR	-	-
Chernyavs'ka and Gullì (2008)	2005-2007	IT	S	Other*	-	>100% if scarcity of capacity	100%
Fell (2010)	2005-2008	SW,FI, DN, NO	S	VECM	-	40-70%	60-95%
Thoenes (2014)	2008-2010	DE	S	VECM	36%	-	-
Jouvet and Solier (2013)	2005-2011	EU	S and F	SEM	>100%	-	-
Freitas and da Silva (2013)	2008-2011	PT	S	VECM	51%	-	-
Hintermann (2016)	2010-2013	DE	S	SEM		111%	81%
Huisman and Kilic (2015)	2007-2013	UK, DE	F	Other**	47-76% (UK), 20-90% (DE)	-	-
Freitas and da Silva (2015)	2008-2013	ES	S	VECM	-	24%	25%

* Load duration curve; ** Kalman filter approach.

Table 3 reports the results of the reviewed literature. While being generally high, estimated PTRs exhibit significant variation. Different factors are behind this. First, different econometric techniques are used. Single Equation Models (SEMs) and, within the framework of co-integration, Vector Error Correction Models (VECMs) are the two most popular approaches. Except for Honkatukia *et al.* (2006), estimated PTRs from VECMs tend to be lower compared to those derived from SEMs.[25] Second, different energy mixes, in different countries, can determine different PTRs. These will be higher if, at the margin of the supply "merit order" curve, electricity prices are set by plants using more carbon intensive technologies. Third, since which plant sets the market price

[25] An additional technical distinction can be made between the studies using electricity forward prices (typically, one-year ahead) and those using spot prices (one-day ahead). The choice of which prices to use for estimation is not neutral. On the one hand, spot prices are more volatile, less driven by fuel or carbon costs and more influenced by unforeseen events such as plant outages or weather shocks. On the other hand, greater data variation is useful for parameter identification (Hintermann, 2016).

also depends on the demand level, most studies distinguish between peak hours and off-peak hours. Given the merit order curve, cost pass-through tends to be higher during peak hours. Also, for some producers, bidding below marginal cost may be convenient to avoid switching off plants for only a few hours (Fabra and Reguant, 2014). Fourth, both market power and excess generation capacity matter (Chernyavs'ka and Gullì, 2008). The more concentrated the market, the higher pass-through is likely to be. Conversely, the more extra capacity available, the lower pass-through is likely to be.[26]

5.2 Manufacturing sectors

Compared to the electricity sector, cost pass-through in manufacturing sectors is more complex to analyse. This is firstly because price information for manufacturing products and for their (potentially many) inputs may not be available or sufficient. Secondly, product heterogeneity implies that, within a sector, market conditions relevant to cost pass-through, such as the degree of international competition, the degree of internal market concentration, transportation costs and spare production capacity, can be very diverse (de Bruyn *et al.*, 2015). Thirdly, some of these conditions may vary across countries and, in international markets, the position of a country along the supply curve may matter too. For these reasons, PTR estimates both are few relative to the number of regulated sectors and can be highly variable across countries.

As far as the methodology is concerned, while all the studies reviewed use the cointegration framework, two approaches can be distinguished with respect to the identification strategy. With the cost-price approach, which is the most common, the price of the given product is explained by the prices of input components (e.g., labour, capital, energy, materials) and by carbon prices. De Bruyn *et al.* (2010a, 2010b)

[26] For the German market, Zachmann and Hirschhausen (2008) find evidence of asymmetric cost pass-through (i.e., electricity prices rise more after carbon price increases than they fall following equivalent carbon price decreases), which is an indication of market power.

introduce the alternative market-equilibrium approach. This approach rests on the assumption that markets are internationally integrated, to the extent that an equilibrium (long-term) relationship exists between domestic and foreign prices. In the typical model, the (domestic) EU price of the given product is explained by the corresponding US price, the carbon price and the exchange rate. The attractiveness of the market-equilibrium approach is in its parsimony. Its potential weakness, on the other hand, is in the assumptions regarding market integration and price adjustments.

Table 4 reports the PTR estimates found in the literature. At a glance, very high PTRs are observed for some sectors, notably the refinery sector and the iron and steel sector, while the opposite is true both for pulp and paper and for the glass sectors. Others exhibit wide PTR ranges depending on the specific products. This is especially the case with the chemicals and with the ceramics sector. As to the cement sector, PTRs vary a great deal across countries. For example, pass-through is high in Portugal and Poland, but low in the UK.

Table 4 - Estimates of carbon/energy cost PTRs for manufacturing sectors.

Study	Approach	Cement	Ceramic	Chemicals	Glass	Iron and steel	Pulp and paper	Refining
Alexeeva-Talebi (2010) [1]	Cost-price	100%(DE)		100%(DE)	0%(DE)		0-38%(DE) [9]	
Alexeeva-Talebi (2011)	Cost-price							100%(EU)
de Bruyn *et al.* (2010a, b)	Market-equilibrium			33-100% (EU) [5]		>100% (N.EU)		>100%(DE)
de Bruyn *et al.* (2015)	Cost-price	20%(FR), 40%(DE), 90-100% (CZ, PL)		0%(N.EU, NL), 100%(EU) [6]	0%(ES, DE), 40%(FR), 60%-100% (IT)	55-100% (EU)		80-100% (IT, BE, FR, DE, GR)
Oberndorfer *et al.* (2010) [3]	Cost-price		30-100% (UK) [4]	50-100% (EU) [7]	0-25%(UK) [8]			50-75% (UK)

Note: (1) Unless differently specified, values refer to electricity PTR. (2) Values refer to energy PTRs, except for Refining. (3) 30% for bricks, 100% for ceramic goods. (4) 33% for PS, 100% for PE and PVC. (5) 0% for two fertilisers, 100% for other fertilisers and petrochemicals. (6) 50% for fertilisers, 100% for PE. (7) 0% energy cost PTR for container glass, 20-25% for hollow glass. (8) Overall labour, material and energy costs PTR: 0% for paper and paper board, up to 38% for household and toilet paper.

The empirical evidence on cost pass-through is now reviewed sector by sector.

Cement

Two relevant studies were found for the cement sector. One by Alexeeva-Talebi (2010), for Germany, analyses pass-through of electricity-, materials- and labour costs during Phase I. Producers are found to pass through up to 73% of cost shocks and, specifically, 100% of electricity cost shocks. De Bruyn *et al.*'s (2015) study covers several European countries during Phases I to III. Depending on data availability, different products are considered, namely Portland cement (by far, the most common type of cement), total cement (i.e., all types of cement) and clinker (a semi-processed product). For Portland cement, almost full pass-through (90-100%) of carbon costs is found in Poland and the Czech Republic. By contrast, for total cement, PTRs range from 20% in France to 40% in Germany. For clinker, estimated PTRs are around 35-40%. Considering the low transportability of cement, estimated PTRs are perhaps not as high as one might expect (except for Poland's and Czech Republic's results). The authors suggest three possible explanations, which relate to production overcapacity, long-term contracts (in the presence of which empirical models cannot properly capture the effect of carbon prices) and the incentive to increase production associated with activity level thresholds.[27]

[27] See the works of Neuhoff *et al.* (2014) and Branger *et al.* (2015) reviewed in Section 4.2.

Ceramics

For the ceramics sector, only one relevant study was found (Oberndorfer *et al.*, 2010). Here, the estimated PTRs refer to energy costs, not carbon costs. Pass-through for ceramic goods and ceramic bricks, in the UK, are analysed. While full pass-through (100%) is found for ceramic goods, a PTR of 30-40% is estimated for ceramic bricks. The differences between the two types of products explain the different results.

Chemicals

The chemicals sector comprises many subsectors and products. The literature mainly focuses on two subsectors: petrochemicals and fertilisers. Concerning the first, different types of plastics are considered, notably polyethylene (PE), polystyrene (PS), polyurethane (PUR), polypropylene (PP) and polyvinylchloride (PVC). As to the second, ammonium products and nitrogen compounds are the goods most frequently analysed.

De Bruyn *et al.* (2015) offer the most comprehensive analysis of cost pass-through for chemicals. The models for the prices of petrochemicals include, as explanatory variables, the naphtha price (naphtha being the main input cost), the price of emission allowances and other potentially relevant variables, such as energy prices, output and stock indices, wages, interest rates and the euro-dollar exchange rate. The estimated PTRs generally exceed 100%. The same approach is used for fertilisers, except for the naphtha price being replaced by the natural gas price. PTRs over 100% are derived for ammonium nitrate (UK), calcium ammonium nitrate (DE), urea (North Western Europe) and urea ammonium nitrate (FR). By contrast, those for ammonia (NWE) and urea (NL) are not particularly significant. Limited transportability is the main explanation provided for the very high PTRs in this sector.

The above results are largely consistent with those of previous studies. Using the market-equilibrium approach, de Bruyn *et al.* (2010a, 2010b) find full pass-through both for PE and PVC. For PS, which has higher carbon content, the resulting PTR is 33%. In Germany, Alexeeva-Talebi (2010) finds full pass-through of electricity costs for most of the chemical products considered (including fertilisers and nitrogen compounds, basic inorganic chemicals, dyes and pigments and plastics in primary forms). In the UK, Oberndorfer *et al.* (2010) find energy cost PTRs of 100% and 50% for, respectively, PE and fertilisers (ammonium nitrate).

Glass

Analyses of cost pass-through for the glass sector are limited by the lack of data. Hollow glass (e.g., bottles, glasses, jars), which makes 50-60% of the EU's total glass production,[28] is the only type of glass for which multiple PTR estimates exist. Estimates for flat glass (25% of the European glass production, mainly supplied to the building and automotive industries) were not found. For hollow glass, de Bruyn *et al.* (2015) find null carbon cost PTRs both for Germany and Spain, a PTR of 40% for France and one of 60-100% for Italy. Alexeeva-Talebi (2010) and Oberndorfer *et al.* (2010) analyse energy costs pass-through, also for hollow glass, in, respectively, Germany and the UK. The first finds null PTR, while the second find a PTR of 0-25% (0% for the more specific product "container glass"). Alexeeva-Talebi (2010) also finds non-significant PTRs for fibre glass and for "other processed glasses".

In general, except for high pass-through found in Italy, the evidence indicates null to low cost pass-through for hollow glass. De Bruyn *et al.* (2015) relate this result to strong international competition and to market conditions whereby the main input (soda ash) is supplied by a small number of companies and demand is also dominated by a few large multinational firms. Furthermore, as carbon cost is a small share of total production costs (about 1% for hollow glass), pass-through is difficult to detect.

[28] The EU is the largest global producer of hollow glass.

Iron and steel

Using the market-equilibrium approach, de Bruyn *et al.* (2010a, 2010b) analyse carbon cost pass-through for hot and cold rolled coil (two representative steel products) in Northern Europe. PTRs over 100% are found for both products. In the follow-up study, de Bruyn *et al.* (2015) repeat the analysis using the cost-price model and extending country coverage. For hot rolled coil, PTRs of 75% and over 100% are found in Northern Europe and Southern Europe, respectively. For cold rolled coil, estimated PTRs are 85% and 55%, in the same order. Different factors underlie this range of high PTRs. On the one hand, transportation costs play a role in limiting imports.[29] On the other, the fall in demand following the economic crisis and expansion of steel production in Asia resulted in lower utilisation rates in Europe's steel production capacity. Under such circumstances, keeping prices competitive becomes even more important, which may explain incomplete PTRs (<100%).

Pulp and paper

Little empirical evidence on cost pass-through in the pulp and paper sector exists. The evidence we have is not specific to carbon costs, which make up less than 1% of total production costs. Electricity costs are instead significant, as they usually represent about 10% of total costs. Alexeeva-Talebi (2010) finds that, in Germany, the paper industry can pass-through only a small fraction, if any, of its cost shocks. A maximum cost PTR (taking together the costs of labour, energy and materials) of 38% is estimated for household and toilet paper, while no pass-through is found for paper and paperboard.

[29] According to Reinaud (2005), considering transportation costs and assuming full pass-through by European producers, carbon prices would need to be around €28/tCO_2 for Chinese steel to compete with marginal steel production in Europe.

Refining

Using the market-equilibrium approach, de Bruyn *et al.* (2010a, 2010b) find carbon cost PTRs in Germany to be over 100% for petrol, diesel and gasoil. These findings are confirmed by the follow-up study (de Bruyn *et al.*, 2015), which uses the cost-price approach. Only, slightly lower PTRs, ranging from 80% to 100%, are found for petrol in Belgium, France, Germany and Italy. Looking at 14 European countries, Alexeeva-Talebi (2011) also finds extensive evidence for full pass-through of carbon costs. In the UK, Oberndorfer (2010) estimates lower PTRs of 75% and 50% for, respectively, petrol and diesel.

The two cross-country studies, namely Alexeeva-Talebi (2011) and de Bruyn *et al*. (2015), stress the roles of low demand price elasticities and of heterogeneous emission intensities. Both unresponsive demands and the small weight of carbon in total fuel production costs (about 2%) clearly explain high PTRs. However, these may be reinforced by differences across countries in emission intensities: if the plant setting the (international) price passes through 100% of its carbon costs, more efficient plants can increase profits by passing through more than 100% of their carbon costs.

6. Conclusions

In a cap-and-trade system, the rules governing allowance allocation fundamentally characterise the instrument and can determine its performance. In the EU ETS, allowance allocation has evolved over time in multiple ways, above all, with respect to the balance between auctioning and free allocation. Compared to the first two trading periods, the allocation rules adopted with Phase III represent a quantum leap. Auctioning has become the default allocation method for the electricity sector and, in general, free allocation is now limited to about 45% of all allowances. Moreover, discretion in allocation decisions has been minimised thanks to the centralisation of the system and to the application of emission efficiency benchmarks.

In the EU ETS, the purpose of free allocation is to safeguard the competitiveness of regulated firms exposed to international competition and, by so doing, to preserve the global environmental effectiveness of the policy instrument. A wide empirical literature (reviewed in Chapter 2) shows that, so far, the EU ETS has not had significant negative effects on regulated firms. Moderate to very modest carbon prices are the main factor explaining this outcome, but no doubt the generous supply of free allowances has played a role too. Thus, strictly speaking, free allocation has met its purpose. Nevertheless, economic efficiency (cost effectiveness) and equity are fundamental criteria for evaluating the working of free allocation.

With regard to efficiency, the most recurring and, hence, least controversial conclusion in the empirical literature, is that the rule used in Phase III for identifying the sectors at risk of carbon leakage is too lenient. This conclusion is mainly explained by *a)* the obsolete assumptions about future carbon prices and by *b)* trade intensity (TI) alone (i.e., independent of carbon costs) being a poor indicator of carbon leakage risk. On the other hand, it has been demonstrated that the application of efficiency benchmarks – while not free from imperfections – is an important efficiency-enhancing feature of free allocation.[30]

On a different question relevant to efficiency, to minimise windfall profits (which implies freeing up resources for other uses as well as avoiding unjustified wealth transfers from consumers to producers), free allocation would ideally be calibrated considering firms' ability to pass through the costs of regulation. However, estimates of cost pass-through rates (PTRs) for regulated industrial sectors are too variable to inform quantitative allocation rules. Furthermore, allocation adjustments do not come

[30] Efficiency benchmarks have also the merit of improving the perceived fairness of free allocation.

without a cost in terms of dynamic efficiency. First, closure provisions are shown to have delayed exit of uneconomic installations. Second, activity level thresholds (ALTs), while reducing windfall profits from overallocation, resulted in greater production and, at least in the case of the cement sector (the only sector analysed) in lower emission efficiency. Both these issues related to closure provisions and ALTs remain in the EU ETS, but the expansion of auctioning has significantly limited their relevance.

As far as equity is concerned, free allocation in the EU ETS has implications for distributional effects between countries, between and within industries, as well as between producers and consumers. The centralisation of the system, with Phase III, has greatly reduced potential distortions in the European market. As some studies show, free allocation based on EU-wide parameters, vis-à-vis cross-country variation in technology and carbon leakage risk, implies some heterogeneity in effective support across countries. On the other hand, probably more significant distributional effects are neutralised by special provisions benefiting lower-income member states, namely the redistribution of 12% of all allowances to be auctioned, the continuation of (transitory) free allocation for the electricity sector and, as of Phase IV, the Modernisation Fund.

The question of cost pass-through is relevant to economic efficiency as well as to equity. Windfall profits from cost pass-through entail wealth transfers from consumers to producers. But, again, the switch to allowance auctioning for the electricity sector has partly solved this issue. Moreover, while free allocation is substantially less generous for the manufacturing sectors off the carbon leakage list, no further differentiation is made among them. Thus, in the context of shrinking free allocation, cross-sectoral differences in cost PTRs matter both in terms of efficiency and equity.[31]

[31] We found no empirical studies assessing, in a comprehensive way, the efficiency and equity implications of the existing two-tier allocation system. Nevertheless, some advocate for the adoption of a multiple-tier system based on differing cost pass-through abilities in different sectors. The objections

The new or amended free allocation rules defined in the reform for Phase IV can be expected to further improve the efficiency of the allocation system. First, the rule for identifying the sectors at risk of carbon leakage will more efficiently identify such sectors by considering carbon emissions intensity (CeI) and trade intensity (TI) simultaneously. Second, allocations will adjust dynamically for output variations exceeding +/- 15%. This means that fast-growing installations will face more limited increases of carbon costs. At the same time, the rule will allow reduction of windfall profits accruing to installations with shrinking output.

The difficulty of accurately quantifying differences in cost pass-through ability, especially in manufacturing sectors (due to limited data availability and market heterogeneity), is clearly the main obstacle to achieving further efficiency in allowance allocation. The TI indicator is an imperfect proxy for cost pass-through (in)ability, which – as we have seen – depends on product- and location-specific characteristics of the market. The reform for Phase IV does in fact make progress in the direction of *a)* considering the CeI and TI indicators for narrower subsectors, and *b)* considering complementary qualitative assessments of abatement potential, market characteristics and profit margins. However, this improvement in the evaluation of cost pass-through ability (more precise and comprehensive as it is) can only result in the addition of sectors or subsectors to the carbon leakage list, not in their cancellation from it.

are that such a system would be largely arbitrary, it would add complexity and it would likely have threshold effects.

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Appendix

The following formula operationalises the allocation of free allowances in Phase III (European Commission, 2011):

$$FA_{i,t} = BEN_i \times HAL_i \times CLCF_{i,t} \times \mathrm{CSCF}_t$$

where:

- $FA_{i,t}$ is the total free allocation of installation i in year t;
- BEN_i is the benchmark for installation i which depends on the product emissions intensity based on the average emissions intensity of the 10% most efficient installations in the EU ETS in 2007–2008;
- HAL_i is the historical production level based on the highest value of the median annual production levels over 2005–2008 or 2009–2010;

- $CLCF_{i,t}$ is the linear reduction factor for installation i. For the installations which are not at risk of carbon leakage, it goes from 80% in 2013 to 30% in 2020; for the installations that are at risk of carbon leakage, it is 1;
- $CSCF_t$ is the cross-sectoral correction factor to ensure that the total free allocation do not exceed the cap;
- $FA_{i,t}$ is reduced by 50%, 75% or 100% if the annual level of production falls below 50%, 25% or 10% of the historical activity level HAL_i.

Chapter 2

Competitiveness effects and carbon leakage

1. Introduction

Whether implemented through carbon taxation or through emissions trading, as with the EU ETS, the main rationale for carbon pricing is its cost effectiveness in reducing emissions (Baumol and Oates, 1971). The experience with the implementation of carbon pricing, however, has generally proved difficult or contentious at the least. Most notably, concerns about the potential negative effects on domestic firms' international competitiveness, due to increased production costs, are a major obstacle to the unilateral use, or to the deeper unilateral use, of carbon pricing. Competitiveness deterioration also implies that the environmental effectiveness of the policy is affected, as in some measure additional emissions will be generated overseas. The mere displacement of emissions, due to their regulation, is called carbon leakage[32]. The EU ETS is no exception in facing these challenges. Their importance is reflected in the history of the EU ETS, its working and the related policy debate.

While the concerns about detrimental effects on international competitiveness and carbon leakage are clearly legitimate, economic theory itself is not unequivocal as to whether such effects are certain outcomes of unilateral carbon pricing. The negative

[32] Competitiveness deterioration is not the only channel through which carbon leakage may occur. If the region adopting more stringent carbon regulation is sufficiently important in terms of demand for fossil fuels, carbon leakage may also result from lower global fossil fuel prices.

impact of tightening environmental regulation on exports or, over the longer term, on industry location is known as Pollution Haven Effect. In contrast with it, the Porter Hypothesis (Porter and van der Linde, 1995) suggests that market-based environmental policies, by spurring new production processes and products, may in fact lead to improved competitiveness. Does empirical evidence generally support the Porter Hypothesis? There is no consensus (Brännlund and Lundgren, 2009; Ambec *et al.*, 2013; Dechezleprêtre and Sato, 2017). At any rate, carbon pricing can be accompanied by specific measures to guard against potentially adverse effects. In the EU ETS, the free allocation of emission allowances is used for just this purpose.

In the last decade, a large econometric literature has accumulated examining evidence of competitiveness effects caused by the EU ETS. Only two studies (Dechezlepretre *et al.*, 2015 and Naegele and Zaklan, 2017), as far as we are aware, directly test for evidence of carbon leakage; that is, for whether emissions increased outside the EU as a consequence of the EU ETS. While the Pollution Haven Effect vs Porter Hypothesis debate permeates this whole literature, the same studies deal in fact with the more specific question of whether the EU ETS net-of-free-allowances has affected businesses. As seen in the previous chapter, free allocation is a central feature of the EU ETS and a determinant of its performance. Its calibration, including the progressive replacement with auctioning, has been at the heart of major regulatory interventions so far. Thus, it is important to understand whether different sectors have suffered any loss of competitiveness due to the EU ETS, despite free allocation, or viceversa, have benefited from regulation, potentially thanks to free allocation.

This paper offers a comprehensive review of the literature at hand. The main goal of the review is to identify robust findings and to qualify some specific results. Unlike previous literature reviews (notably, Martin *et al.*, 2016), we emphasise the specific or more general relevance of estimation results for sectors, countries and years. We deem that by making these distinctions as clearly as possible, better weighted and hence more accurate conclusions can be drawn based on the different outcomes. Organised in this way, the existing empirical evidence may better inform policy. Moreover, it may make more obvious where empirical evidence is limited or missing.

The first part of the review covers several works that, taken together, consider a range of economic performance indicators (e.g., exports, profits, sales, number of employees and productivity measures) and employ various econometric approaches. As Dechezleprêtre and Sato (2017) point out, there is, as yet, no single accepted test or measure of the competitiveness effects of environmental regulation, so the literature continues to use a variety of outcome measures linked to competitiveness. The second part of the review covers, by contrast, a rather homogenous body of literature that specifically infers investors' appraisal of the influence of the EU ETS on firm competitiveness. It does so by analysing the effects of changes in the price of emission allowances, namely European Emission Allowances (EUAs), on returns on company stock prices. The EU ETS can be valued either as a net extra cost or as an opportunity for increased profits. The sector's and firm's characteristics, and the extent of free allocation, will determine which of these two. Unlike other competitiveness indicators, which measure realised outcomes, stock returns reflect investors' expectations. For this reason, we refer to the first as conventional indicators and to the second as unconventional ones. As most of the works looking at stock returns are quite recent, only a few of them have been covered in the previous literature reviews.

The chapter is organised as follows. Section 2 reviews the literature considering conventional competitiveness indicators and the two studies testing for carbon leakage. Section 3 reviews the literature analysing stock returns. Section 4 discusses the findings and concludes.

2. Effects on conventional competitiveness indicators and evidence of carbon leakage

The econometric literature assessing the competitiveness effects of the EU ETS is very diverse with respect to the variables used for measuring economic performance. It is also diverse with respect to the data, the methods and both the time and country coverage of the applications. With reference to the identified studies that consider conventional competitiveness indicators (in the sense specified), the first part of this section illustrates the choices of the authors in each of these respects. For expository convenience, the two studies found that directly test for carbon leakage (Dechezleprêtre *et al.*, 2015 and Naegele and Zaklan, 2017) are considered together with this literature.

Detailed accounts of the contents and the results of the same literature are subsequently provided[33]. The studies are presented according to whether the respective estimated effects are *a)* sector-generic, country-generic, *b)* sector-generic, country-specific, *c)* sector-specific, country-generic, or *d)* sector-specific, country-specific[34]. We deem that the distinction between sector-generic and sector-specific effects is expedient for policymaking. As understood in the rules for free allocation, the EU ETS legislation needs to carefully consider the different effects that the system has on different sectors[35]. This literature review shows, however, that the studies providing estimates of sector-specific effects are relatively few and together only cover few sectors.

2.1 Mapping the literature: dependent variables, levels of analysis, methods, coverage

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[33] A broader discussion of the evidence and of the relative implications of this literature is left to the chapter's conclusive section.

[34] We do not consider effects referring to the manufacturing sector as such as being sector-specific, since the manufacturing sector is too heterogeneous. However, effects referring to any manufacturing subsectors (distinguished by output) are considered sector-specific.

[35] Arguably, heterogeneous effects across countries are easier to evaluate. Moreover, differently from business representatives, national governments participate first hand in the legislative process.

The econometric applications in the literature presented in this section can vary from one to another across different dimensions. The following five are the main ones:

1) Measure of competitiveness

Several dependent variables are considered as alternative competitiveness indicators. This is possible because firms can respond to carbon pricing in different ways, notably through pricing-, production- and investment decisions, the effects of which, in turn, are detectable at different levels. These include sales, profits, the number of employees, exports, process or product innovations and changes in production capacity or location[36].

Table 1 – Surveyed literature (conventional competitiveness indicators): measures of competitiveness.

Variable	Number of studies
Productivity	7
Trade	7
Employment	6
Profitability	4
Turnover	3
Value added	3
Investment (FDIs included)	3
Production costs	1

2) Level of analysis

Thanks to the increasing availability of micro-level data, most applications are at the installation level or, more often, at the firm level. Basic information on regulated installations is found in the EU Transaction Log (EUTL).[37] Additional useful

[36] Dechezleprêtre and Sato (2017) put some order into this collection of indicators by classifying the corresponding effects as first, second and third order, and distinguishing between firm responses, on the one hand, and economic, technology, international and environmental outcomes, on the other.

[37] The EUTL records official information both on regulated installations and allowance transactions in the EU ETS. The EUTL is accessible online at: http://ec.europa.eu/environment/ets/.

information can be drawn from external company databases (typically, private databases provided under license or public ones for administrative purposes) using firm identifiers. Fewer analyses are carried out at the sector level, which typically use time-series techniques.

3) Approach

Different approaches are used. The difference-in-differences (DiD) approach (with pre-treatment matching) has become by far the most popular. Its main attractiveness lies in the clearcut causal interpretation of the results around the impact of participation in the EU ETS – which in the control-treatment framework is the "treatment" – on the given dependent variable. Moreover, it allows for the capture of the final net effect of the EU ETS without having to specify a relationship (which is empirically difficult to identify) between EUA prices and the dependent variable[38]. In the remaining applications, either time-series or panel-data models are employed. In the first case, the effect of the EU ETS is more specifically that of EUA prices. In the second, the effect usually refers to participation in the EU ETS, to EUA prices, or some other measure of regulatory stringency.

Table 2 – Surveyed literature (conventional competitiveness indicators): approach and level of analysis.

	Level of analysis		
Approach	Firm (/plant)	Sector	Total
Time series	0	4	4
Panel data	3	2	5
DiD	12	0	12
Total	13	5	21

[38] Typically, in the studies using the DiD approach, regulated firms or installations are first paired through statistical matching (e.g., through propensity score matching) with similar non-regulated counterparts. The model for the variable of interest is then fitted, including a binary variable among the regressors. This indicates whether the firm or installation participates in the EU ETS. The estimated coefficient of this variable measures the estimated effect of the EU ETS. For a technical overview of this and similar methods for policy evaluation, the reader is referred to Imbens and Wooldridge (2009).

4) Specificity of effects

Some models provide estimates of sector- and/or country-specific effects, whereas others offer estimates of more general average effects. Importantly, models fitted to data from multiple countries or sectors do not necessarily provide estimates of multiple country- or sector-specific effects. While including binary variables controlling for country or sector fixed effects, they often limit themselves to estimating more general average effects that correspond to the scope of the sample.

Table 3 – Surveyed literature (conventional competitiveness indicators): specificity of effects.

	Sector dimension		
Country dimension	Sector-specific	Sector-generic	Total
Country-specific	5	4	9
Country-generic	7	5	12
Total	12	9	21

5) Time coverage

Models are estimated using data spanning different time periods. This is particularly relevant considering the dynamic nature of the EU ETS, as reflected in the evolution of EUA prices, in its structure by trading period, as well as in the changes to its regulation. Most studies are based on datasets covering Phase I (2005-2007) or both Phases I and II (2008-2012). By contrast, surprisingly few studies, as of yet, draw on data stretching as far as Phase III (2013-2020).

Table 4 – Time coverage of surveyed literature (conventional competitiveness indicators).

Study	Phase I			Phase II					Phase III			
	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Reinaud (2008)	■	■	■									
Commins *et al*. (2011)	■	■	■									
Abrell *et al*. (2011)	■	■	■	■								
Costantini and Mazzanti (2012)	■	■	■									
Sartor (2012)	■	■	■	■	■	■	■					
Chan *et al*. (2013)	■	■	■	■	■							
Yu (2013)	■	■										
Petrick and Wagner (2014)	■	■	■	■	■	■						
Wagner *et al*. (2014)	■	■	■	■	■	■						
Lundgren *et al*. (2015)	■	■	■	■								
Dechezleprêtre *et al*. (2015)*			■	■	■	■	■	■	■	■		
Marin *et al*. (2015)	■	■	■	■	■	■	■	■				
Jaraite and Di Maria (2016)	■	■	■	■	■	■						
Branger *et al*. (2016)	■	■	■	■	■	■	■	■				
Klemetsen *et al*. (2016)	■	■	■	■	■	■	■	■	■			
Lutz (2016)	■	■	■	■	■	■	■	■				
Borghesi *et al*. (2018)	■	■	■	■	■	■						
Koch and Basse Mama (2016)	■	■	■	■	■	■	■	■	■			
Löschel *et al*. (2016)	■	■	■	■	■	■	■	■				
Boutabba and Lardic (2017)	■	■	■	■	■	■	■	■	■	■	■	
Naegele and Zaklan (2017)*			■				■					

* Study directly testing for carbon leakage.

2.2 Detailed review

2.2.1 Sector-generic, country-generic effects

Using a large firm-level longitudinal dataset, mainly sourced from the Amadeus global company database (Bureau van Dijk), Commins *et al.* (2011) assess the effects of energy taxes and of the EU ETS on different competitiveness indicators. Four panel data models are estimated for firm-level: *a)* employment; *b)* tangible investment; *c)* total factor productivity (TFP); and *c)* return on capital (ROC), over 1996-2007. In each model, the effect of the EU ETS is captured by a binary variable, indicating whether the firm's main business is in one of the regulated sectors. Participation in the EU ETS is found to have negative effects both on TFP (-3.2%) and ROC (-4.7%). Conversely, the effect on employment is positive (1.5%). All these effects, however, are statistically significant only at the 10% level. Moreover, since participation in the EU ETS is defined at the sector level[39], measurement error is incurred and sectoral shocks may also confound estimations of the effects. The authors warn that the results are indicative.

Costantini and Mazzanti (2012) estimate a sector-level gravity model of international trade for manufacturing exports from 15 EU countries to 145 importing countries, over 1996-2007. The effect of the EU ETS is captured by a binary variable for the years 2005-2007, which is the period corresponding to Phase I. Its estimated coefficient indicates that the EU ETS increased exports of medium-low technology sectors which roughly correspond to those covered by the EU ETS: thus, an outcome consistent with the Porter Hypothesis. The authors themselves, however, qualify this finding as being far from conclusive, stating that further sector disaggregation and longer time series are required to infer on the impact of the EU ETS on the competitiveness of regulated firms.

Using firm-level data and applying the DiD approach, Marin *et al.* (2015) estimate the effects of the EU ETS on a large set of competitiveness indicators: value added; number of employees; turnover; investment; labour productivity; wages; return on investment (ROI); TFP; and markup. As in other studies in this literature, the Amadeus database is the main data source, providing information on the EU ETS firms and on matched counterparts. The samples used for estimation comprise between approximately 700 and 800 regulated firms (or fewer depending on any missing values of the variable analysed), from 2002 to 2012. The EU ETS appears to have positively influenced the scale of firms, as measured by turnover, employment and value added

[39] In a given sector, some firms participate in the EU ETS, while others do not. Notably, firms operating installations whose capacity is below the relevant threshold do not come under the EU ETS.

(14.9%, 8.2% and 6%, respectively, in Phase II). By contrast, negative effects, smaller in magnitude, are found for two scale-free variables, namely -2.4% and -0.5% for, respectively, TFP and ROI, in Phase II. Some heterogeneity in effects across sectors and firms is also identified. The interactions with the treatment variable denoting participation in the EU ETS indicate that emission-intensive firms and sectors were generally penalised. On the other hand, consistent with the Porter Hypothesis, positive effects on labour productivity and on ROI are found for the interaction between participation in the EU ETS and (patent-based) environmental innovation.

2.2.2 Sector-generic, country-specific effects[40]

Wagner *et al.* (2014) assess the impacts of the EU ETS on GHG emissions and employment of regulated manufacturing firms in France. The DiD approach is applied to panels of administrative data comprising 5,957 plants (384 regulated) and 4,589 firms (287 regulated), from 2000 to 2010. Sizeable effects on emissions are found for Phase II, which, however, differ depending on whether the analysis is conducted at the plant or at the firm level. In the first case, the authors find that the EU ETS reduced emissions by almost 20%; in the second, by half as much. Similarly, varying results are found for the effects on employment. At the plant level, Phase II turns out to have reduced employment by 7%; at the firm level, meanwhile, no statistically significant effects are found. Two possible explanations for these differences in results are: within-firm leakage, that is, firms shifting production from regulated to non-regulated plants;

[40] Anger and Oberndorfer (2008) estimate the impact of relative allowance allocation, this being the ratio of allocated allowances to emissions, on revenues and on the employment of German regulated firms in 2005. The perspective is, therefore, slightly different from that of the studies surveyed here, which consider the effects of participation in the EU ETS, and not of relative allocation among regulated firms. The results indicate that relative allocation had no statistically significant effect on revenues or employment.

and measurement error[41]. The authors find evidence of only modest within-firm leakage, but warn that further testing is required.

Petrick and Wagner (2014) estimate the impact of the EU ETS on CO_2 emissions and CO_2 intensity, as well as on employment, gross output (sales) and the exports of regulated manufacturing firms in Germany. The DiD approach is applied to firm-level panel data obtained from the national production census, covering over 90% of the EU ETS installations operated by manufacturing firms and located in Germany. About 400 regulated firms are in the estimation samples. In the reference model, the effects on employment are not statistically significant, while positive effects on the value of both sales and exports are identified for the first three years of Phase II (i.e., 2008-2010)[42]. The positive effects are clearly non-robust, however, as they become statistically insignificant in most of the alternative estimations performed.

Jaraite and Di Maria (2016) estimate the impact of the EU ETS on CO_2 emissions and CO_2 intensity, as well as on the investment and the profitability of Lithuanian regulated firms. The DiD approach is applied to a panel dataset sourced from the national business survey, of almost 5,000 firms (about 330 of which are regulated), from 2003 to 2010. Investment and profitability are measured by the change in total tangible capital assets and the ratio of before-tax profit to turnover, respectively. It turns out that, up to 2009, regulated firms invested, on average, less than non-regulated firms, but conversely, in 2010, they invested more. This may well be the consequence of a Lithuanian law passed in 2009, mandating that all revenues received from the sale of emission allowances should be spent on environmental measures. Negative effects on profitability, though statistically significant only at the 10% level, are detected in 2009 and 2010. This may be explained by allowance allocation in Phase II being much tighter for Lithuanian firms than in Phase I.

[41] Participation in the EU ETS is defined at the installation level, not at the firm level.

[42] The data do not allow for the disentangling of quantity and price variations.

Klemetsen *et al.* (2016) estimate the impact of the EU ETS on CO_2 emissions and CO_2 intensity, as well as on value added and labour productivity (value added per man hour) of Norwegian-regulated manufacturing plants[43]. The DiD approach is applied to a plant-level panel dataset obtained by combining different datasets, the most important of which is compiled by the Norwegian Environment Agency. The sample eventually used for estimation comprises 152 plants (72 regulated), from 2001 to 2013. Very large positive effects, though only for Phase II (when carbon prices were higher), are found both on value added and labour productivity: respectively, 24% and 26%. The authors relate these results to generally non-stringent allowance allocations and, also, to the pass-through (to output prices) of the connected opportunity costs.

Borghesi *et al.* (2018) investigate whether Phases I and II had any effect on outward FDIs of regulated manufacturing firms in Italy. In this study, FDIs are measured by the number of foreign subsidiaries[44]. The DiD approach is applied to a longitudinal dataset extracted from the Aida database (Bureau van Dijk) of Italian companies, from 2002 to 2010. The samples used for estimations include treatment groups of about 300 regulated firms. Results show that, on average, firms in the EU ETS did not increase their presence in other countries. However, the interactions with the treatment variable, denoting participation in the EU ETS, reveal that firms operating in sectors particularly exposed to international competition increased their FDIs towards countries not covered by the EU ETS. This result may reflect a strategy by which regulated firms attempt to stay competitive in the market.

Similarly to Borghesi *et al.* (2018), Koch and Basse Mama (2016) test for whether the EU ETS affected FDI stocks outside the EU of regulated (almost exclusively, manufacturing) firms in Germany[45]. The DiD approach is applied to a panel of firm

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[43] Norway joined the EU ETS in 2008.

[44] Available information on the economic relevance of the subsidiaries is too scarce to be used.

[45] The analysis only considers firms that have invested abroad.

survey data (Deutsche Bundesbank) spanning 1999-2013 (within which 232 regulated firms are identified). When using the full sample, no statistically significant effects are found across the three trading periods. This is the primary result, which speaks to exaggerated concerns about carbon leakage through relocation. Subsample analysis, however, reveals that the EU ETS increased (by as much as 50%) FDI activity of firms in "non-process" regulated sectors (notably, the machinery, electrical equipment and automotive sectors). Compared to "process" regulated sectors, non-process ones are characterised by lower capital intensities[46]. For this reason, the latter are more geographically mobile.

2.2.3 Sector-specific, country-generic effects

Abrell *et al.* (2011) have the merit of being the first to use the DiD approach in an application to the EU ETS. The study draws on a large dataset extracted from the Amadeus company database. The dataset covers over 2,000 regulated firms (operating about 3,600 installations), from 2003 to 2008. Separate models for CO_2 emissions, added value, employment and profit margin are fitted. Average impacts of the EU ETS on regulated firms are found to be not statistically significant, with the exception of a small negative effect (0.9%) on employment over 2004-2008. Some heterogeneity in estimated average effects arises when the models are fitted to widely-defined sector sub-samples[47]. Notably, the profit margins of firms in the energy sector (electricity and heat) benefit from participation in the EU ETS. By contrast, firms producing non-metallic mineral products (e.g., glass, cement, ceramics, bricks) are negatively affected. The authors warn that the results should be interpreted with caution: firstly, because

[46] "Process" sectors are those regulated through process-specific capacity thresholds. The "non-process" sectors are regulated due to combustion activities.

[47] Four sector subsamples are considered: "Paper and paper products", "Non-metallic mineral products", "Basic metals" and "Electricity and heat".

firms in the treatment and control groups (while similar with respect to a number of variables) operate in different sectors; secondly, because the potential indirect effects (of the EU ETS through electricity prices) on firms in the control groups are ignored.

The study by Chan *et al.* (2013) is similar to Abrell *et al.*'s (2011) both in terms of the data used and approach. The main methodological difference is that matched firms, in the treatment group and in the control group, here belong to the same sector[48]. As to the scope of the analysis, three sectors are examined: electricity; cement; and iron and steel. For each of them, three competitiveness indicators are considered: unit material cost (i.e., the ratio of total material costs to turnover); employment; and turnover. Statistically significant effects are found only for the electricity sector, which exhibits increases both in unit material cost (by 5% and 8% in Phases I and II, respectively) and, more substantially, in turnover (by 30% in Phase II). The authors conjecture that the increase in material costs may be related to compliance costs (i.e. the cost of purchasing allowances) or fuel switching from coal to gas. The larger increase in turnover would reflect the carbon cost pass-through to output prices.

Branger *et al.* (2016) and Boutabba and Lardic (2017) deal with the impact of the EU ETS on the competitiveness of the cement and the steel industries. Both contributions model EU27's net imports of the two products applying time-series regression techniques. In Branger *et al.* (2016), both the effects of EUA prices on net imports of cement and of steel are not statistically significant. Boutabba and Lardic (2017) revisit Branger *et al.*'s (2016) analysis applying a rolling cointegration approach, which accounts for multiple structural changes, and uses longer data series updated with more recent information. The EUA prices are found, for both cement and

[48] This excludes the risk of estimated effects being biased by different trends in the sectors to which the firms in treatment and control groups would belong. On the other hand, it risks firms which should be in the treatment group ending up in the control group. Abrell *et al.* (2011) warn that this can happen if participating firms are not identified in the company database.

steel to have (time-varying) effects on net imports that are positive and statistically significant for some subperiods. The authors interpret the results as suggesting that modest operational leakage took place, and that it was more evident in the steel sector than in the cement sector.

Using similar approaches to those of the two aforementioned studies on steel and cement, Reinaud (2008) and Sartor (2012) investigate the impact of the EU ETS on the competitiveness of the primary aluminium sector. Direct emissions from (both primary and secondary) aluminium production came under the EU ETS only in 2013, with the start of Phase III. However, as primary aluminium is an electricity-intensive product (electricity represents over a third of total production costs), the impact of the EU ETS on the sector is principally felt indirectly through electricity prices[49]. In Reinaud's (2008) preliminary regression of the EU's net imports of primary aluminium, only the first two years of the EU ETS are covered. She finds the effect of EUA prices not to be statistically significant. The main explanation offered is that electricity used in primary aluminium production was most often provided under long-term electricity contracts, shielding aluminium producers from rising electricity costs. Sartor (2012) extends Reinaud's (2008) analysis, using longer data series, which reach 2011 (by when many electricity contracts were expected to have expired), and controlling for both additional variables and cointegration. The findings, nevertheless, confirm those of the previous study. They suggest that other factors are much more important than the carbon price in determining the competitiveness of the primary aluminium sector: energy prices, electricity contracts and other factors driving electricity prices, as well as exchange rate movements.

[49] In the primary production process, aluminium oxide is produced from bauxite and further processed to aluminium via electrolysis. Secondary aluminium is produced using recycled scrap. Its electricity intensity is about 5% of that required to produce primary aluminium.

2.2.4 Sector-specific, country-specific effects

Yu (2013) estimates the impact of the EU ETS on profit margins (ratio of net profits to the turnover) of Swedish firms in the energy sector (electricity production, electricity distribution, steam and hot water supply), in 2005 and 2006. The DiD approach (with and without matching[50]) is applied to a panel dataset, extracted from Statistics Sweden's business database, comprising almost 1,000 firms (113 regulated), from 2004 to 2006. The estimation results do not show a statistically significant impact for the EU ETS on firm profitability in 2005. However, a negative significant impact (1.1 percentage point decrease in the profit margin ratio) is found for the following year. The results are tentatively interpreted in light of carbon prices and machinery investments (a proxy for abatement investments). Notably, an increase in machinery investment is observed in 2006 only for regulated firms.

Lundgren *et al.* (2015) assess the effects of the EU ETS (as well as of the Swedish energy and carbon taxes) on the TFP of Swedish pulp-and-paper sector firms. A firm-level panel dataset (supplied by Statistics Sweden) is used, of some 100 firms over 1998-2008. Their approach involves two steps. Firm-specific Luenberger indicators of TFP growth are first computed[51], distinguishing between: *a)* efficiency change; and *b)* technological development. In the second step, the impact of the EU ETS carbon price on TFP growth is estimated, using a dynamic panel data approach. Interestingly, carbon prices are found to have a positive effect on efficiency change and, conversely, a negative one on technological development. The latter result is in line with those of Commins *et al.* (2011) above.

[50] The authors state that the results with matching are similar, but they are not shown in the paper (they are available from the authors upon request).

[51] The Luenberger indicators are computed based on directional distance functions using data envelopment analysis, which is a non-parametric linear programming technique.

Lutz (2016) estimates the impact of the EU ETS on the TFP of regulated manufacturing firms in Germany. Using firm-level administrative data comprising about 15,000 firms (of which about 400 are regulated), for the period 1999 to 2012, industry-specific production functions are first estimated and firm-specific productivity levels are then derived. The DiD approach, with and without matching, is then applied. Positive effects on productivity are detected. Specifically, for Phase I, 0.7% and 1.5-2.7%, without and with matching, respectively; and for Phase II, 1.2-1.4%, only with matching. Moreover, subsample analysis (presented only for DiD without matching) reveals some heterogeneous effects across sectors: a positive one (2.4%) for basic metals in Phase I; while no statistically significant effects are found for the food, paper, and chemical industries. The author warns, however, that, since the productivity measure employed is revenue-based, the results may at least in part reflect pass-through of regulation costs.

Using data and methods similar to those employed by Lutz (2016), Löschel *et al.* (2016) investigate whether the EU ETS had any impact on production efficiency of regulated manufacturing firms in Germany. In this study, the firm-level distance to the estimated sector-specific production frontier is the relevant measure of economic performance[52]. Across the array of modeling strategies, no evidence is found of negative effects of the EU ETS on efficiency. On the contrary, the EU ETS is found to have had a positive impact for firms in the paper sector (-1.3% and -1.6% over 2003-2007 and 2003-2012, respectively) and, in general, on regulated firms during the first year (only) of Phase I.

2.2.5 Carbon leakage

[52] A negative effect represents a move toward the efficient production frontier (i.e., an increase in production efficiency).

Dechezleprêtre *et al.* (2015) are the first to directly test for carbon leakage caused by the EU ETS. That is, instead of testing for competitiveness effects and assuming that those would have resulted in carbon leakage, shifts in emission locations are directly analysed. An ingenious strategy is put to work, made viable by a special database. The Carbon Disclosure Project (www.cdp.net) collects climate-relevant data at the firm level, including on the emissions of multinational firms broken down by country. As they already operate from multiple locations, multinational firms are believed to be the most prone to shift production activity across countries and, hence, to carbon leakage. Using a sample covering 1,785 companies (142 regulated), over 2007-2014, the authors test the relationship between changes in firms' European share of their own emissions and changes in their extra-EU emissions share. Existence of carbon leakage would imply that such relationships would be negative. No negative effects are found. The authors note, however, that region-specific productivity shocks could potentially confound the estimated effects of the EU ETS.

Naegele and Zaklan (2017) investigate whether the EU ETS caused carbon leakage in European manufacturing sectors, as measured by changes in sector-level international trade flows and related carbon movements. Sector-level trade flows in embodied carbon and value are computed using detailed trade and input-output data (from the Global Trade Analysis Project) for the years 2004, 2007 and 2011. Two models are estimated, namely for net imports and for bilateral flows (thus allowing for intra-industry trade) and four alternative measures of environmental stringency are considered for representing the EU ETS. The simplest of these measures is an indicator variable for regulated sectors, while the others are (sector-specific) "direct"-, "indirect"-, and "total net" costs based on emissions (direct/indirect), allowance allocations and allowance prices. As no significant effects are found, the authors conclude that, during its first two phases, the EU ETS did not have a systematic impact on flows of trade or embodied CO_2 emissions.

3. Effects on stock returns

Taking a radically different approach from the literature reviewed in Section 2, several studies analyse the competitiveness effects of the EU ETS through the lens of stock markets. Assuming that a firm's stock price reflects its stream of expected discounted future profits, the effect of changes in EUA prices on stock price returns reveals investors' beliefs about the influence of the EU ETS on profitability. Specifically, a direct (/inverse/no) relationship between carbon prices and stock prices indicates that investors see the EU ETS as having a net positive (negative/no) effect on profits. Crucially, this effect varies across sectors and firms depending on the respective characteristics and on the proportion of emissions covered by free allowances. Indeed, the typical exercise in this literature consists in testing for differing effects across such dimensions.

Considering the mechanisms underlying the Porter Hypothesis, but also more sophisticated ones (e.g., restricting entry, raising rivals' costs), the impact of carbon pricing on profits is not predictable with any certainty. In this sense, free allocation in a cap-and-trade system adds a layer of complexity. Free allocation introduces two channels through which firms can increase revenues and by which it can potentially increase profits; the latter meaning that the revenue increase following carbon pricing exceeds the production cost increase. The first channel is the pass-through of opportunity costs (the costs associated with the use of free allowances for compliance) to output prices. With positive carbon prices, depending on its ability to pass-through (at no expense of market shares or profits) and on its endowment of free allowances relative to emissions, a firm increases revenues and possibly profits. The second channel refers to the extra revenues that can be obtained from selling unused allowances. Thus, an increase in carbon prices will have a positive (/negative) effect on profits if the revenue effects operating through the said two channels dominate (/are dominated by) the increase in production costs. By the same token, a decrease in carbon prices will have a negative (/positive) effect on profits if the negative revenue effects dominate (/are dominated by) the decrease in production costs. Any of these variations in carbon prices equally indicates that investors see the EU ETS as having a net positive (/negative) effect on profits.

The literature in question presents itself as quite homogeneous in methodological terms. The use of the multifactor market model (MMM) is typical, in which daily (or monthly) returns on a firm's stock are explained by those returns on the market portfolio and on other prices deemed to be relevant. These include carbon prices and fuel prices (e.g., coal, oil, gas). The basic MMM framework, applied to panel data, has the following form:

$$R_{it} = \alpha + \sum_{j} \alpha_i D_i^j + \beta_1 R_t^m + \beta_2 R_t^c + \sum_{j} \gamma_j R_t^j + \varepsilon$$

(1)

where: R_i is the return on the stock of firm *i*; D_i is a firm dummy variable (taking the value 1 when *i* = *j*); R^m is the return on the market portfolio; R^c is the return on emission allowances; and the R^js are the returns on fuel prices.

Most often daily returns are considered, while only a few studies use monthly data[53]. The number of companies varies greatly, from a dozen to about one hundred. More relevant elements of differentiation concern the estimation of effects that vary across sectors, firms or trading periods. Early contributions tend to focus on the electricity sector, within which they distinguish dirtier from cleaner firms (defined according to the carbon intensity of the generation fuel mix). More recent studies tend

[53] Monthly data can be preferable as there is less noise in their information. However, the tradeoff is with the number of available observations, which, of course, are many fewer than with monthly data.

to extend the analysis to other sectors and can draw on data covering the first years of the third trading period. This is important, given the fundamental change in the allowance allocation regime brought in with Phase III. Moreover, the most significant variations on the standard MMM approach are represented by event studies (exploiting the April 2006 crash of EUA prices) and the extension to vector co-integration.

In the following, the identified literature (Table 5) is described in greater detail. The studies looking exclusively at stocks of companies in the electricity sector are first considered. We then turn to those extending the analysis to multiple sectors.

Table 5 – Time coverage of surveyed literature (stock market).

Study	Phase I			Phase II					Phase III			
	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Oberndorfer (2009)												
Veith *et al.* (2009)												
Mo *et al*. (2012)												
Bushnell *et al.* (2013)												
Jong *et al*. (2014)												
Scholtens *et al*. (2014)												
Oestreich and Tsiakas (2015)												
Venmans (2015)												
Pereira da Silva *et al.* (2015)												
Tian *et al.* (2016)												
Pereira da Silva *et al.* (2016)												
Moreno and Pereira da Silva (2016)												

3.1 Electricity sector (exclusively)

Oberndorfer (2009) and Veith *et al.* (2009) are the first to analyse the effect of changes in EUA prices on stock returns. The two studies present similarities beyond the MMM approach. Though the samples differ somewhat in the number of companies (12 and

22[54], in Oberndorfer and in Veith *et al.*, respectively), their time spans largely coincide (August 2005 – June 2007 and April 2005 – August 2007, in the same order). Above all, the analyses share the main outcome: on average, changes in carbon prices and in stock prices were positively correlated. Deeper analyses offer additional results and insights. Notably, Oberndorfer (2009) finds that effects varied by time and by country (where companies have their headquarters). On a different level, Veith *et al.* (2009) find that increases in carbon prices did not have a positive effect on carbon-free generation companies[55].

Mo *et al.* (2012) mainly extend the analysis of previous works by comparing estimated effects across trading periods. The estimation sample covers the years 2006 to 2009, thus straddling Phases I and II. The authors find that, on average, correlation between changes in carbon prices and in stock prices was positive in the first trading period (as in both Oberndorfer, 2009 and Veith *et al.*, 2009) and, by contrast, it was negative in the second. Moreover, stock prices were much more sensitive to changes in carbon prices in Phase II than they were before. In both cases, differences are attributed to more stringent allowance allocations in Phase II.

Further variations on the theme are provided by Tian *et al.* (2016) and by Pereira da Silva *et al.* (2016). The former, who can rely on a sample covering Phases I and II in full, show that the relationship between carbon prices and stock prices was largely driven by (two) specific shocks in the carbon market. Outside these special periods, the relationship depended on the carbon intensity of electricity producers: negative for carbon-intensive companies, positive for less carbon-intensive ones. Limiting the analysis to the Spanish stock market, but crucially drawing on data straddling Phases II

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[54] Perhaps, even 22 companies may seem a small number to some. However, those considered by Veith *et al.* (2009) represent almost two thirds of total electricity generation in Europe and account for about one third of all emissions under the EU ETS.

[55] This is unexpected as, even in the short term, carbon-free generators benefit from higher electricity prices by reaping inframarginal rents.

and III, Pereira da Silva *et al.*'s (2016) extend the standard MMM approach to vector cointegration. Applying a cointegrated Vector Error Correction Model (VECM), which allows for a control on dynamic interactions among variables, the authors find that: the equilibrium relationship between carbon prices and stock prices was (weak, but) positive in Phase II; and statistically non-significant in Phase III. This result fits well with the switch from free allocation to full auctioning for the electricity sector. Moreover, the above positive relationship is found to be significantly stronger for the subset of clean energy producers.

3.2 Multiple sectors

Bushnell *et al.*'s (2013) study stands out in this literature for adapting the standard MMM approach to the form of an event study, as well as for the depth of analysis. The abrupt April 2006 fall in EUA prices, following the disclosure of 2005 verified emissions[56], had immediate repercussions on the European stock market. Focusing on a three-day window (April 26-28), the authors examine the changes in stock returns, across sectors and firms, to elicit investors' beliefs about the influence of the EU ETS on firms' profitability. As a result of the crash in carbon prices, stock prices fell for firms in energy-intensive sectors, particularly for those selling primarily within the EU. In the electricity sector, price stocks of less carbon-intensive producers were more affected than those of dirtier ones. Data variation is not sufficient for identifying an effect of allowance holdings (emissions and allowances are highly correlated), but findings are unaffected when allowance holdings are considered. The main conclusion of the study is that, under free allocation[57], investors focused on increasing profits through higher output prices. Analysing the same market event and using a similar

[56] On 25 April, the first reports on country-level emissions began to leak into the market. As the excess supply of allowances became obvious, carbon prices fell from €28/tCO_2 on 25 April to €14 on 28 April.
[57] In Phase I (and in subsequent Phase II), almost all allowances were given away for free.

approach, Jong *et al.* (2014) find evidence confirming the stronger negative impact of the fall in carbon prices on dirtier firms.

Also the studies not restricted to the energy sector mostly use the standard MMM approach. Prominent among them is Oestreich and Tsiakas (2015). Using monthly data from the German stock market, the authors provide compelling evidence on the role of the allocation method in affecting the relative financial performance of different firms. The comparison between the average excess returns of "dirty" firms, who received the largest amounts of free allowances, and those of "clean" ones, who did not receive any, is unequivocal. Namely, before 2005, no difference can be detected; then, dirty firms significantly outperformed clean firms, but only up to March 2009: when the second ETS Directive stipulated that allowances would be mainly auctioned beginning in 2013. After March 2009, clean firms outperformed dirty ones.

Venmans (2015) contributes firm and sector-specific estimates of correlations between changes in EUA prices and in stock prices. These refer to all companies in the StoxxEurope Total Market Index belonging to EU ETS sectors, over the period 2007-2011. Mean effects are positive for all seven sectors under consideration. The least important effects are found for the electricity, refining and the paper sectors; intermediate effects are found for the cement and chemicals sectors, whereas the iron and steel and the nonferrous sectors present the largest effects. However, the study emphasises that effects are firm specific and that variation within sectors can be significant. Out of 97 firms, 19-24 (depending on the model) exhibit negative effects. Within the electricity sector, less carbon-intensive firms, as expected, benefit more from carbon price increases.

Focusing on the Spanish stock market, Pereira da Silva *et al.* (2015) apply the VECM variant of the MMM approach to firms in five different sectors: electricity; refining; iron and steel; pulp and paper; and, finally, the cement, ceramics and glass sectors taken together. The estimation sample is short in time, as it only covers (and not in full) the first two years of Phase III (2013-2014). Estimated effects are positive for all sectors, except for the iron and steel and for the pulp and paper sectors; here they are, respectively, negative and statistically not different from zero. Using the standard MMM approach, Moreno and Pereira da Silva (2016) extend the analysis by including full Phase II and by adding companies in the chemicals sector. Effects are shown to be both sector and phase-specific. Unlike other studies, negative effects are found for the electricity sector. The only sector for which positive effects are found in both trading periods is the refining sector[58].

4. Conclusions

The empirical literature on the competitiveness effects of the EU ETS and related carbon leakage is wide and growing. Yet, only two studies were found directly testing for carbon leakage; that is, testing for whether emissions increased outside the EU as a consequence of the EU ETS[59]. This is the case principally because econometric identification of carbon leakage presents additional empirical challenges (greater data requirements to begin with) compared to identification of competitiveness effects. Within the reviewed literature, two main sets of works have been identified. The first considers a range of conventional indicators measuring economic outcomes linked to competitiveness, such as profits, exports, sales, employment and productivity. The second looks at the stock market to infer whether investors believe the EU ETS is beneficial or detrimental to profits. Though the respective approaches are very different, the two bodies of literature complement each other nicely. Their recurring findings are

[58] The refining sector is characterised by very high cost pass-through rates, close to 100% (see, e.g., de Bruyn *et al.*, 2010 and de Bruyn *et al.*, 2015).

[59] None of these applications can produce empirical estimates of carbon leakage rates (i.e., the share of locally abated emissions compensated by increased emissions overseas). To do this, one would need a structural model allowing for the simulation of a counterfactual world without carbon pricing (Aichele and Felbermayr, 2015).

largely consistent, thus, allowing for some robust conclusions. Nevertheless, especially considering the information needs of policymakers, significant gaps remain.

By far, the most frequently encountered conclusion is that no evidence was found of negative statistically significant effects of the EU ETS on firms' competitiveness (nor, therefore, of carbon leakage). Moderate to very low carbon prices are the first obvious suspect for this outcome. However, the role of generous free allocation, especially considering firms' differing abilities to pass through opportunity costs of emission allowances, is not less important. A few negative effects have been found, but most of them are characterised by uncertainty greater than ideal (i.e. estimated effects are statistically significant only at the 10% level) or are accompanied by words of caution due to specific data or methodological issues. Besides, a large diversity in the competitiveness indicators deployed means that a particular variable for which negative effects are more frequently detected does not exist. Nevertheless, while any significant effects should be carefully considered, two recent studies prompt a warning that, perhaps, should receive special attention. Namely, Borghesi *et al.* (2018) and Koch and Basse Mama (2016), both find evidence of greater FDI activity, respectively, in Italy's and Germany's regulated sectors most exposed to international competition and less capital intensive.

Positive effects are more frequently found, but, crucially, they are almost exclusively explained by windfall profits (whether via pass-through of carbon costs, allowance overallocation or inframarginal rents) rather than by increased productivity and larger market shares. Indeed, it remains unclear whether the EU ETS made firms in any sectors or subsectors more competitive through innovation or efficiency improvements, the mechanisms underlying the Porter Hypothesis[60]. By contrast, some

[60] Some studies notably, Calel and Dechezleprêtre (2015), find that the EU ETS had significant positive effects on low-carbon innovation (for a survey of the relevant literature, see Rogge, 2016). However, whether these resulted in net gains in competitiveness is a different question.

of the studies analysing stock returns are particularly effective in showing that the combination of free allocation and the pass-through of opportunity costs resulted in profit increases. They do so through comparisons of estimated effects: *a)* between regulated and non-regulated sectors; *b)* between dirtier and cleaner producers within the electricity sector; and *c)* across the first two trading periods and Phase III, Phase III marking the switch from almost full free allocation to auctioning as the main allocation method.

Finally, the conclusions that can be drawn from this literature present three orders of limitations, all of which are, at least in part, explained by empirical difficulties. First, most empirical analyses still only refer to Phases I and II. This delay might have some relevance considering the changes that have taken place since the start of Phase III, notably the changed allocation regime. But developments external to the EU ETS (e.g., the economic context, technology, climate policies in other world regions) need also to be taken into account. Second, distinct estimates of sector-specific effects are few. With reference to the literature considering conventional competitiveness indicators, we found: three estimates for the cement sector; three for the iron and steel sector; three for the energy sector (for which carbon leakage risk is minimal); two for aluminium; two for paper; and one or, more often, none for all other sectors[61]. While scarce in number, sector-specific estimates seem to us particularly valuable, as they are suited to inform policy, not least possible revisions of the EU ETS. Third, with few exceptions, existing estimates of competitiveness effects refer only to short-term effects. However, the most relevant economic losses that unilateral carbon pricing might cause would unfold over the long term, if expected or protracted competitiveness deterioration affected investment decisions. In other words, through industry relocation or what is commonly referred to as investment leakage.

[61] In some cases, two-digit NACE sectors are still too heterogenous for the corresponding estimates to be of real use for policy.

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Chapter 3
Interactions with other policies

1. Introduction

The ultimate goal of climate mitigation policy is the stabilisation of atmospheric greenhouse gas (GHG) concentrations "at a level that would prevent dangerous anthropogenic interference with the climate system" (United Nations, 1992). Considering the pervasiveness of GHG emissions in virtually all human activities, each with its own technical and socio-economic specificities, the use of composite policy approaches for achieving said goal is justified or just necessary. The ubiquity of emission sources also implies that policy instruments conceived and adopted for combating climate change are most likely to interact with other pre-existing policy instruments. Ideally, the components of a policy mix complement each other, create synergies and, thereby, reduce the cost of achieving the given objective(s). By the same token, however, negative interactions within a given set of policies may result in higher costs. This is an increasingly important policy issue in view of the major mitigation efforts that need to be made. The design and implementation of cost-effective policy mixes has indeed become one of the most debated topics in the field of climate policy, especially in relation to emissions trading.

Any policy that can induce a significant reduction in GHG emissions can be considered climate policy. A basic distinction is, then, between "explicit" and "implicit" climate policy instruments. "Explicit" instruments are specifically conceived for reducing GHG emissions and, as such, they explicitly target emissions. "Implicit" instruments, by contrast, are instruments whose application can result in significant emission reductions but this is not their primary purpose. Examples of explicit climate policy instruments are the two possible means for establishing carbon pricing, namely carbon taxes and emission trading systems (ETSs)[62]. Explicit climate policy instruments are also emission standards for vehicles or appliances as well as for other forms of taxation (not carbon taxes) calibrated on emission intensity, such as the vehicle registration taxes or the circulation taxes in use in some countries. Examples of implicit climate policy instruments are numerous too. A range of instruments are used for promoting renewable energy sources (RESs), such as feed-in tariffs, feed-in premia, green certificates, tendering schemes, but also investment subsidies, both in the forms of direct payments and tax credits. Other instruments specifically incentivise energy efficiency (EE): these include energy efficiency standards, white certificates and, again, investment subsidies[63]. Excise taxes on petrol, or on other energy goods, while conceived and used for raising public revenues, can also have a central role in limiting emissions. Furthermore, as a result of differing objectives and policy stratification over time, instruments or measures working in the opposite direction of climate policy instruments continue to exist. We here refer to subsidies (including those in the form of exemptions on energy excise taxes) that effectively incentivise the production or use of fossil fuels over RESs or EE.

The specific purpose of emissions trading is to minimise the total cost of achieving a given emission reduction target. Thus, a higher cost that may be incurred because of other concomitant policies is a particularly relevant issue. Accordingly, among the several types of interactions and evaluation criteria that might be

[62] ETS is here used as a synonym of cap-and-trade system.

[63] For comprehensive surveys on the use of RES and EE support instruments in Europe, see Kitzing *et al.* (2012) and CEER (2013).

considered[64], the way other climate policies may affect the cost-effectiveness of the EU ETS is the focus of our analysis. Under the static short-term perspective, climate policies overlapping with the EU ETS increase the total cost of reducing regulated emissions. This is generally the case because overlapping policies imply that marginal abatement costs differ across regulated firms. Under the dynamic long-term perspective, overlapping policies put downward pressure on the price of emission allowances (by reducing the demand for emission allowances), thus weakening the economic incentives for investment in innovation and adoption of low-carbon technologies.

In the EU, all above-mentioned instruments are variously implemented across member states and sectors. The European climate policy landscape is, therefore, complex. Within the EU's wider strategy aiming to reconcile the goals of competitive energy prices, energy security and environmental sustainability, the EU ETS is the main instrument in use for decarbonising the economy and – crucially – it is the only major EU-wide climate policy instrument in operation. As explained above, however, the efficiency of the EU ETS is not independent of a multitude of mostly national RES and EE policies implemented for pursuing the respective goals. The strong impact of the economic crisis on the EU ETS also had the result of relaunching the debate on the influence of all these policies on the system and, more generally, on the need for a coherent and, hence, efficient policy mix. The establishment of the Market Stability Reserve (MSR), within the EU ETS, is the reform with which problems posed by concomitant RES and EE policies, or for that matter by any factor weakening the carbon price signal, has been centrally addressed.

The importance of the issue at hand is reflected in the wide literature dedicated to it. Sizeable research projects have been undertaken or commissioned by various

[64] See, for example, the theoretical frameworks laid out by del Río González (2007) and Oikonomou and Jepma (2008).

institutions. Only in the last few years, the EU has financed two projects focusing on the interactions between climate and energy policies, namely ENTRACTE[65] and CECILIA[66], which in turn followed the INTERACT project (Sorrell *et al.*, 2003). The International Energy Agency (IEA) has also devoted significant efforts to analysing the same policy interactions, the result being a number of reports and articles (e.g., Hood, 2011, 2013 and Philibert, 2011). Crucially, however, most studies only provide ex-ante analyses, given both the relatively short experience with climate policy mixes and the empirical challenges that ex-post analyses entail, especially in this domain. Empirically assessing the differences in costs between alternative policy combinations is, indeed, anything but simple, as it requires estimating credible counterfactual scenarios. Moreover, almost all ex-post studies deal specifically with the interactions between the EU ETS and RES-E policies, that is, policies supporting electricity from RESs.

The rest of the chapter is organised as follows. Section 2 explains the rationales for using multiple climate policy instruments. Section 3 outlines the setting of climate and energy policies in the EU. Section 4 illustrates the MSR. Section 5 reviews the relevant literature on the interactions between the EU ETS (or more generally carbon pricing) and other policies. Section 6 concludes.

2. Rationales for carbon pricing and additional policies

Most economists favour carbon pricing over alternative approaches to climate mitigation on the grounds that it can cost-effectively reduce emissions (Baumol and

[65] www.entracte-project.eu

[66] www.cecilia2050.eu[67] The six legislative texts are the following: the Directive revising the EU ETS (European Parliament and Council, 2009a); the Decision setting national targets for CO_2 emissions from the non-ETS sector (European Commission, 2009a); the Directive setting national targets for RES in the energy mix (European Parliament and Council, 2009b); the Directive creating a legal framework for carbon capture and storage (European Parliament and Council, 2009c); the Regulation setting standards for CO_2 emissions from new cars (European Commission, 2009b); and the Directive revising the Fuel Quality Directive (European Parliament and Council, 2009d).

Oates, 1971). Whether in the form of carbon taxation or – conditional on certain assumptions – emissions trading, uniform carbon pricing is cost-effective in that it equalises marginal abatement costs across all subject polluters. Carbon pricing is cost-effective for other reasons as well, including, among others, the continuous incentive to further reduce emissions (in an intertemporal setting) and its modest information requirements (Baranzini *et al.*, 2017). Nonetheless, it is an equally widely shared view that, over the long term, carbon pricing alone is not sufficient to induce major emissions reductions at minimum cost. Some complementary policies are required, particularly to promote innovation and appropriate infrastructure investment (Bowen, 2011). Exactly which complementary policies should be adopted is, though, a debatable question.

In the case of an ETS, such as the EU ETS, possible negative interactions with other policy instruments are particularly salient. Notably, the application of additional (explicit or implicit) climate policies in any of the sectors already covered by the system implies that the marginal cost of abatement (i.e. the cost of reducing an additional tonne of carbon) is not homogenous across all regulated sectors. This means that not all the least costly abatement opportunities are exploited first, contrary to the very purpose of emissions trading. That is, the way that abatement is achieved is not cost-effective. To give a realistic example, RES policies result in greater supply of renewable energy than would be obtained, at the same total cost, under an ETS alone. In the second case, any abatement opportunities cheaper than RES investments would first be exploited, such as fuel switching or some measure of energy demand reduction. Moreover, RES as much as EE policies overlapping with an ETS reduce the demand for emission allowances and, thereby, put downward pressure on carbon prices. Lower carbon prices reduce compliance costs, which per se is not a bad thing. However, low carbon prices may affect the dynamic efficiency of the system; that is, its ability to incentivise low-carbon innovation and investment and, therefore, to minimise abatement costs over the long term.

Despite these implications, there are some good reasons for using multiple climate policy instruments in addition to carbon pricing (see, e.g., Sorrell and Sijm, 2003; Sijm, 2005; Bennear and Stavins, 2007; Levinson, 2012; Lehman and Gawel, 2013; Löschel and Schenker, 2014). First, multiple market failures, additional to the emission externality, justify the use of multiple instruments. In the EE domain, market failures include information failures, principal-agent problems (as with the landlord-tenant setting) and transaction costs (Linares and Labandeira, 2010). These and other factors, such as those related to bounded rationality, lead to sub-optimal EE investment levels. On the producer's side, the most relevant market failure concerns knowledge spillovers. The basic idea is that firms underinvest in activities generating knowledge because they cannot fully appropriate the returns to it. In economic terms, the private rate of return is lower than the social rate of return. Different types of knowledge spillovers exist, notably: *a)* learning-by-searching and *b)* learning-by-doing and learning-by-using. As learning-by-searching stems from R&D investment, publicly-funded research is the corresponding typical policy response. Learning-by-doing and learning-by-using depend on, respectively, cumulative experience in producing a product and cumulative experience in using a product. Greater production is expected to increase productivity and, hence, to reduce the average cost per unit, as experienced in the production of photovoltaic (PV) modules (IRESA, 2016). Learning-by-using is especially relevant to technologies that have complex and interdependent components (Kamp, 2007). Again, the PV and wind industries offer some examples. As in this case the result is underuse of the technology, support to technology deployment, whether in financial or regulatory forms, is the corresponding policy response.

Second, a basic principle in economic policy, known as the Tinbergen rule, is that for each and every policy target there must be at least one policy instrument. Failing this condition, some policy goals will not be achieved (Tinbergen, 1952). By definition, the primary purpose of any implicit climate policy instrument is different from reducing emissions, though crucially its use has a significant effect on emissions. Notably, both RES and EE policies are essential for improving the security of energy supply. They are also central in the development of the domestic green industry. These goals, while different from climate mitigation, justify the use of RES and EE policies in addition to carbon pricing.

Third, bringing all emission sources under an ETS is generally considered to be uneconomic because of the dispersion of small emitters and related high administrative costs. Though in theory different ETS designs exist (see, e.g., Ellerman *et al*., 2006), even the technical viability of these options is not, in fact, obvious. For these reasons, existing ETSs do not cover 100% of all emissions – in fact, they usually come nowhere near – in their respective economies. Complementary instruments are, thus, needed to reduce emissions in the sectors left uncovered. A convenient way of doing this is to use carbon taxation. Carbon pricing is cost-effectively extended across the economy if the rate of the complementary carbon tax matches the price of emission allowances or is at least sufficiently close to it. In the EU, a few member states (e.g., France, Ireland, among others) apply carbon taxes in the non-ETS sector (which covers transportation, the residential sector, agriculture and the non-energy intensive industries) in connection with expected EUA prices.

Fourth, additional policies are sometimes used for tackling regulatory or political constraints (Löschel and Schenker, 2015). Policies may be adopted for mitigating the opposite effects of pre-existing ones that prove difficult to remove. For example, the removal of fossil fuel subsidies is a primary reform for cost-effectively reducing emissions. However, because of the difficulties intrinsic to this type of reform, it is plausible that RES support would be preferred as a more feasible alternative. Similarly, additional policies may be adopted for contrasting the negative effect of political uncertainty on the effectiveness of carbon pricing. From an investor's perspective, the fact that current policymakers wish to keep carbon pricing may not be a sufficient guarantee for its actual future. This may cause underinvestment in low carbon technologies, despite the presence of carbon pricing, and, thus, motivate additional policies or measures.

3. Climate and energy policy in the EU

3.1. The 2020 horizon

All explicit and implicit climate policies currently in force in the EU are framed within the 2020 Climate and Energy Package (C&EP). Approved in 2009, the C&EP consists of six legislative texts[67] that, together, lay out a set of targets to be achieved by 2020. There are three main targets, namely: *a)* 20% reduction in GHG emissions from 1990 levels; *b)* 20% share of RES in final energy consumption; and *c)* 20% reduction in total energy consumption relative to a business-as-usual (BAU) scenario. Unlike the first two targets, the third, which is the one relevant to EE, is not legally binding.

The EU ETS is the principal instrument for reaching the GHG target. However, since it covers less than half (about 45%) of the EU's overall emissions, the EU ETS is accompanied by targets and related instruments that are applicable in the non-ETS sector. For this ad-hoc defined sector, which encompasses all economic sectors other than electricity generation and energy-intensive manufacturing, the Effort Sharing Decision (ESD) sets national emission reduction targets. These targets, which are differentiated on the basis of GDP per capita, range between -20% (from 2005 levels)

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[67] The six legislative texts are the following: the Directive revising the EU ETS (European Parliament and Council, 2009a); the Decision setting national targets for CO_2 emissions from the non-ETS sector (European Commission, 2009a); the Directive setting national targets for RES in the energy mix (European Parliament and Council, 2009b); the Directive creating a legal framework for carbon capture and storage (European Parliament and Council, 2009c); the Regulation setting standards for CO_2 emissions from new cars (European Commission, 2009b); and the Directive revising the Fuel Quality Directive (European Parliament and Council, 2009d).

for the wealthiest member states and +20% for the least wealthy[68]. Collectively, the ESD targets impose a 10% reduction in the EU's emissions. Contrary to what happens in those sectors covered by the EU ETS, no major EU-wide policy is in place for reaching the ESD goals. According to the subsidiarity principle, it is the responsibility of member states to devise and implement the necessary policies and measures.

Turning to the RES target, just as with the ESD, member states are assigned individual targets, differentiated on the basis of GDP per capita, but also depending on the initial RES share (at the time of the decision) and on the potential for increasing such share. National targets differ widely, ranging from 10% (Malta) to 49% (Sweden). Member states are responsible for the development of the policies supporting accelerated RES deployment and for the prioritisation of any sector involving transportation, buildings, and heating and cooling. The only constraint is a minimum 10% share of RES in transportation, to be achieved through biofuels (mainly) and the diffusion of electric and hydrogen cars. Various instruments promoting RES production, such as feed-in tariffs, green certificates, tendering schemes and investment subsidies, have been implemented by all member states. Along with these schemes, facilitating measures have been adopted, such as priority dispatch and exemptions for small plants from burdensome administrative procedures. The result has been a significant increase in RES installed capacity and production, especially in the electricity sector.

As to the third and final EE target, apart from a few EU-wide regulations, like those concerning the eco-design requirements for energy-related products and the labelling of electric appliances, member states have to define and implement national policies for reducing energy consumption. Concerns about the real possibility of missing the EE target caused the EU to subsequently adopt the Energy Efficiency

[68] The countries allowed to increase their emissions still have to adopt mitigation policies, as their targeted emission levels are lower than projected BAU levels.

Directive (European parliament and Council, 2012). The Directive more precisely quantifies the overall EE target (still for 2020, still non-binding). It defines a common framework for promoting EE in the EU and it sets binding targets in specific areas, notably renovation of public buildings and metering for residential electricity and heat consumption.

3.2 Beyond 2020

Climate and energy policies require a vision of the future and long-term planning. In 2009, after the adoption of the C&EP and shortly before the UNFCCC conference in Copenhagen, the European Council expressed its ambition to reduce GHG emissions by 80-95% from 1990 levels by 2050. Consistent with this political commitment, the European Commission presented the 2050 Roadmap for a low-carbon economy (European Commission, 2011a) and the 2050 Energy Roadmap (European Commission, 2011b). These roadmaps were followed by a Green Paper (European Commission, 2013), the adoption of the 2030 Climate and Energy Framework (C&EF) (European Commission, 2014a) by the European Council, the launch of the Energy Union (European Commission, 2015a) and the Clean Energy Package (European Commission, 2016). Further, given the changed context since the previous 2050 Roadmap for a low-carbon economy, the European Commission recently presented the EU long-term vision for a climate-neutral economy by 2050 (European Commission, 2018).

The structure of the 2030 C&EF closely resembles that of the 2020 C&EP. After intense debate, it was decided to keep the three goals separate, keep the same baselines, and strengthen the targets to be attained by 2030 (Buchan and Keay, 2015; Rossetto, 2015). The GHG target is a 40% reduction in emissions (from 1990 levels), again to be achieved through both the EU ETS and national policies for the non-ETS sector.[69]

[69] Here, the inclusion of land use and land use change and forestry is a significant novelty.

Overall emissions from the non-ETS sector must be cut by 30% relative to 2005 levels. Concerning the RES target, this is a 27% share of final energy consumption at the EU level. Unlike before (under the C&P), however, it is not broken down by member state. The target will be achieved through coordinated national initiatives. As to EE, a collective and only indicative 27% reduction of energy consumption relative to BAU projections is foreseen.

The will to pursue a comprehensive approach to European climate and energy policies has become all the more visible with the Energy Union (European Commission, 2015a). All the relevant policy streams are gathered under this strategy for the purposes of improving their coherence, reinforcing the different but closely-related initiatives and, ultimately, bringing greater energy security, sustainability and competitiveness to European citizens and firms. Two of the five dimensions of the Energy Union are particularly relevant to the topics addressed in this chapter: "Energy efficiency contributing to moderation of demand" and "Decarbonising the economy".

Following the path presented in the Energy Union, the European Commission has undertaken several legislative initiatives that will reshape the policy landscape for the period 2021-2030, and which are part of the Clean Energy Package (European Commission, 2016). Importantly, the RES and EE targets of the C&EF have been raised further by the Clean Energy Package. The two relevant pieces of legislation are the revised Renewable Energy Directive (European Parliament and Council, 2018b) establishing a EU target of at least 32% share of renewable energy consumption for 2030 and the revised Energy Efficiency Directive (European Parliament and Council, 2018c) setting a 2030 EU target of at least 32.5% reduction of energy consumption relative to BAU. Moreover, the new Governance Regulation (European Parliament and the Council, 2018d) includes the requirement for member states to draw up integrated National Energy and Climate Plans for 2021 to 2030 outlining how to achieve the targets and submit the draft to the European Commission by the end of 2018.

4. The Market Stability Reserve

By the start of Phase III, the EU ETS had accumulated a surplus of about two billion allowances.[70] To give an idea of the significance of this supply and demand imbalance in allowances, two billion tonnes of CO_2 (equivalent) is more than the total volume of annual emissions under the EU ETS. According to the most recent data available, the surplus reduced, in 2016, to just below 1.7 billion allowances (European Commission, 2017). The economic crisis, which peaked in 2009, was the main cause of the initial fall in allowance demand. The persisting surplus originated from this major event, combined with the perfect rigidity of allowance supply, which was a fundamental feature of the EU ETS. As expected, the allowance surplus severely depressed EUA prices. This is problematic in so far as the incentives for low-carbon innovation and investment are weakened, which means the dynamic efficiency of the system may be compromised.

In 2012, the European Commission started tackling the problem by adopting a short-term measure known as "backloading". As the name suggests, backloading consisted in postponing the auctioning of 900 million allowances within Phase III, from 2014-2015 to 2019-2020. However, given the long-lasting nature of the surplus, which was expected to increase up to 2.6 billion allowances in 2020 (European Commission, 2014b), further action proved necessary. The Market Stability Reserve (MSR) was then established. The MSR consists in a rule-based mechanism to cope with possible shocks

[70] Surplus is defined as the difference between the cumulative amount of allowances available for compliance at the end of a given year, and the cumulative amount of allowances effectively used for compliance with the emissions up to that given year (European Commission, 2014c).

to allowance demand (European Commission, 2015b). By introducing some flexibility in allowance supply, the MSR is intended to mitigate the impacts on the carbon market of macroeconomic shocks and, for that matter, of any unforeseen event or development affecting allowance demand, including the deployment of RES and EE technologies.

With the MSR, the number of allowances to be auctioned depends partly on the market surplus, as measured by the "total number of allowances in circulation" (see footnote above):

- if surplus exceeds 833 million allowances, allowances equal to 12%[71] of the surplus are withheld from auctions and added to the reserve;
- if surplus is lower than 400 million, 100 million allowances are taken from the reserve and injected into the market through auction;
- if surplus is anywhere between 400 and 833 million allowances, no intervention is triggered.

The thresholds triggering the adjustments to allowance supply delimit an interval of surplus values within which "experience shows that the market was able to operate in an orderly manner" (European Commission, 2014b). To determine the trigger values, operators' hedging needs are the key variable to consider (European Commission, 2014c).[72]

While the MSR started operating in January 2019, the first reduction in allowance supply will only take place in 2021. This two-year lag in the functioning of the MSR has a technical reason: emissions from year *t-1* are verified in mid-*t*, thus allowing the corresponding supply adjustment only to occur in *t+1*. In connection with the start of

[71] This percentage rises to 24% in the period from 2019 to 2023.

[72] Typically electricity generators need to cover at least part of their forward sales with emission allowances.

the MSR, however, it was eventually decided that the 900 million allowances already withdrawn through backloading would not be re-injected into the market at the end of Phase III, as had been originally planned. Instead, they will be moved to the reserve. The European Commission will monitor the operation of the MSR, which will, then, be formally reviewed every five years.

With the entry into force of the new ETS Directive in April 2018 (European Parliament and Council, 2018a), the Market Stability Reserve (MSR) has been substantially reinforced. According to Article 2 of the new ETS Directive, between 2019 and 2023 the feeding rate, i.e. the amount of allowances put in the reserve, will double to 24% of the allowances in circulation. The former feeding rate of 12% will be restored as of 2024. Another important amendment to the MSR is that (unless otherwise decided in the first review of the MSR, in 2021) from 2023 onwards the number of allowances held in the reserve will be limited to the auction volume of the previous year through invalidation of those in excess. This new mechanism will be operational as of 2023.

The main question about the MSR concerns its real impact on the carbon market. In theory, assuming that the market surplus will eventually be reabsorbed (given the indefinitely declining cap) and under unlimited banking (as is the case in the EU ETS), but also conditional on the assumptions of perfectly forward-looking agents and the absence of both market failures and regulatory uncertainty, the MSR would have no effect on carbon prices. In practice, however, since not all these assumptions perfectly hold in reality, the MSR is expected to have some effects on carbon prices (Neuhoff *et al.*, 2015). Reductions in allowance supply would be followed by price increases, and vice versa (i.e. supply expansions would be followed by price decreases), thus smoothing out the pattern of carbon prices over time.

Estimating the impact of the MSR on carbon prices is a very difficult task, mainly because it depends on the expectations and the behaviour of heterogeneous agents. The

Impact Assessment by the European Commission (European Commission, 2014c) recognises this difficulty and, indeed, it does not provide annually-detailed estimates of the impact of the MSR on prices. The difficulty of the task is implicitly proven by the studies that offer such estimates, as these turn out to be strongly dependent on the assumptions of the models that the authors use. While some studies conclude that the MSR will improve the functioning of the EU ETS, others show that the MSR may, in fact, have a negative impact and increase price volatility[73]. In this regard, the trigger parameters are an essential feature of the MSR design. If trigger levels are too high, the MSR will be ineffective; if they are too low, they may create volatility (Neuhoff *et al.*, 2015).

Furthermore, some analysts have emphasised the potential vulnerability of the MSR (in its ability to partially insulate carbon prices from demand shocks) to any significant deviation from the expected scenario. This includes the unanticipated deployment of RES and EE technologies. For example, in a counterfactual scenario exercise carried out by Thomson Reuters,[74] carbon prices are expected to reach, in 2030, about €25/tCO_2. However, holding the current MSR parameters, carbon prices are forecast to stay well below €15 if the EU's EE 2030 target is raised to 40% or in a scenario of EU-wide coal phase out. Thus, the analysis confirms that the periodic review of the MSR trigger parameters (something that the legislator has already foreseen) is a necessary institutional arrangement for effectively countering possible unanticipated effects of other climate policies.

[73] The reader is referred to the special issue of the Journal of Environmental Economics and Management dedicated to the economics of the MSR (Hepburn *et al.*, 2016).

[74] https://blogs.thomsonreuters.com/answerson/future-coal-europe-dark-carbon-prices-take-cue/

5. Literature on the interactions between carbon pricing and other policies

Over the past two decades, a large economic literature has developed on the interactions between carbon pricing, emissions trading in the first place, and other climate and energy policies. Within the same literature, however, a clear imbalance is observed between the numerous studies conducting ex-ante analyses, on the one hand, and the many fewer empirical studies assessing the interactions of existing policies, on the other. Ex-post analyses, indeed, often present additional technical challenges (principally related to data availability) and they can, necessarily, only consider policies that have been in place for a sufficiently long time. The first part of this section outlines the ex-ante literature on climate policy interactions in the European context. Thereafter, a detailed review is provided of the corresponding ex-post literature that is relevant to the EU ETS.

5.1 Ex-ante analyses

The ex-ante literature can roughly be divided into three groups. The first group includes theoretical models, often complemented with numerical applications (e.g. Fischer and Newell, 2004; Pethig and Wittlich, 2009; Böhringer and Rosendahl, 2010). This literature is useful for understanding the dynamics of policy interactions. The second group uses computable general equilibrium (CGE) models to study the dynamics of policy interactions across heterogeneous sectors (e.g. Böhringer *et al.*, 2009; Boeters and Koornneef, 2011; Flues *et al.*, 2014). These studies are often used for estimating the medium and long-term impact of new policies. The third group focuses, instead, on the power sector, typically using partial equilibrium models with detailed data on generation (e.g. Palmer and Burtraw, 2005; De Jonghe *et al.*, 2009). The results of all this ex-ante literature strongly depend on model assumptions and the data used. Moreover, they usually have difficulty incorporating and assessing market failures. In general, they show that GHG pricing tends to be the most cost-effective way to reduce emissions. However, in the presence of market failures (particularly knowledge spillovers for renewable technologies), an energy policy portfolio can reduce emissions more efficiently than a single pricing option. In this case, subsidies for R&D are usually considered the best complementary policy.

The results of ENTRACTE confirm that carbon pricing is generally the most cost-effective option for reducing emissions. However, in the presence of other externalities linked to learning-by-searching and learning-by-doing, the use of additional instruments can significantly reduce the costs of abatement. Schenker *et al*. (2015) show that a portfolio of polices addressing these externalities can reduce compliance costs by a third compared to a scenario with carbon pricing as the only climate policy instrument. However, there are at least two reasons for which it is difficult to design a policy portfolio that successfully addresses those externalities: first, policymakers may not have sufficient information; second, the policy mix needs to be dynamic, adding another layer of complexity. As the externalities get smaller over time (since knowledge spillovers tend to diminish), also the scope of complementary instruments needs to be reduced in order to adapt to the new situation. Hence, introducing additional policies should be done carefully.

CECILIA 2050 showed that the mix of EU-level instruments and unilateral national policies in conjunction with the lack of co-ordination among them, causes differences in abatement incentives and produces different abatement costs between sectors and member states (CECILIA, 2014). Nevertheless, the EU policy mix has proved effective in reducing emissions and, Meyer and Meyer (2013) show, the climate and energy policy mix may have produced net economic benefits for the EU. Further, the concept of "optimality" is studied in the context of the policy mix. Optimality does not only include the criteria of effectiveness (*i.e.*, whether the policy mix reaches the target) and cost-effectiveness (at what cost), but also of "feasibility", from a political, legal, and administrative perspective. It is concluded that an optimal climate policy mix is difficult to achieve, but improvements are possible.

5.2 Ex-post analyses relevant to EU ETS

Within the large and growing empirical literature on the EU ETS, little more than a handful of studies assess the interactions between the EU ETS and other policies, typically RES-E policies. Within this limited literature subset, three main questions (related to one another) are addressed. First, some works estimate the extent to which observed reductions in emissions from the electricity sector, or from all sectors covered by the EU ETS, are due to complementary policies, as opposed to the EU ETS itself. Second, some studies focus on estimating the costs of abating CO_2 emissions through the deployment of different RES technologies. Third, (only) one study, by Koch *et al.* (2014), assesses the impacts of market "fundamentals" related to marginal abatement costs (including, among others, expectations on economic activity and RES-E deployment) on the dynamics of EUA carbon prices. Below, the literature review is organised according to said three research questions.

5.2.1 Abatement from complementary policies

Weigt *et al.* (2013) estimate the impacts of the EU ETS and of RES deployment (in each case, with and without the other instrument) on CO_2 emissions in Germany's electricity sector, 2006-2010. The counterfactual analysis is carried out using a unit commitment model, which is a plant-level partial equilibrium model of the electricity system, allowing simulation of electricity dispatch. The authors calculate that without RES, but holding historical (observed) carbon prices, emissions from the electricity sector would have been 11% to 20% higher over the period analysed. In the opposite case, namely without the EU ETS (a scenario modelled by setting carbon prices equal to zero), but holding historical RES capacity, emissions would have been only 1% to 3% higher. It is, thus, concluded that RES policies were much more effective than the EU ETS in reducing emissions. Moreover, RES generation is found to have been more effective in reducing emissions than it would have been in the absence of the EU ETS (i.e. zero carbon prices). Though the difference is small, as it ranges between 0.5 and 1.5% of emissions, this is an interesting positive interaction effect between the two policies[75]. Using a similar approach to Weigt *et al.*'s (2013), Van den Berg *et al.* (2013) estimate the impact of RES deployment on CO_2 emissions from the electricity sector of 12 member states in Western and Southern Europe, 2007-2010. The main simulation results show that without RES injections due to support schemes, emissions of said electricity sector would have been 10% to 16% higher, over this period.

An entirely different type of analysis, as far as the empirical approach is concerned, is carried out by Meyer and Meyer (2013). Using GINFORS, which is a multi-country, multi-sector macroeconometric model of the global economy, the authors estimate the individual impacts of the UE ETS and of RES policies (in the electricity sector) on CO_2 emissions, GDP and employment levels. They do this for each and every member state, 1998-2008. The model is first calibrated to the historical values of key macroeconomic variables, so as to derive the historical baseline scenario against which the counterfactuals of interest can be subsequently evaluated. Without the EU ETS, it is found, 2008 emissions would have been 1% to 3% higher in most member states. The corresponding positive effects on economic activity (GDP) and

[75] The quantity of emissions abated through RES injections (into the grid) depends on which plants, at the margin of the merit order, are displaced. In the electricity system analysed, the interaction effect turns out to be positive because, with carbon pricing, coal plants are more often at the margin of the merit order than they would be otherwise.

employment, however, would have been significantly smaller. Turning to RES policies, two counterfactuals are considered. In both RES capacity is fixed at 1998 levels, thus implicitly assuming that all subsequent RES deployment has been policy-induced. In one scenario, however, RES investment is replaced by fossil fuel investment, while keeping total investment unchanged; in the other, RES investment is also replaced by fossil fuel investment but (more realistically) total investment is reduced. Under the first scenario, Germany, Portugal and Spain would have had, respectively, 9%, 7.8% and 5.6% more CO_2 emissions in 2008, respectively. These are the largest variations, while the average change across member states is 2.5%. The corresponding positive effects on economic activity and employment are small, as total investment is unchanged. Without RES investment and with reduced total investment, results mainly change with respect to economic activity and employment (as one would expect), for which negative variations clearly prevail. It is, thus, concluded that RES policies had positive effects on emission reductions and, also, that they probably had positive effects on GDP and employment in most European countries.

Gloaguen and Alberola (2013) econometrically assess the relative importance of different factors in explaining emissions from the sectors covered by the EU ETS, 2005-2011. A panel-data model, which includes the share of RES in electricity generation, economy's energy intensity, economic activity, as well as energy and carbon prices among other explanatory variables, is, first, fitted to country-level annual emissions (all members are included in the estimation sample). The fitted model reproduces the historical baseline scenario sufficiently well and is, thus, used for performing counterfactual simulations. The authors calculate that reductions of around 1.1 billion tonnes of CO_2 have been achieved within the scope of the EU ETS. More than half of these reductions (600-700 million tonnes) resulted from RES and EE policies. Significant, but substantially smaller, are the impacts of the economic downturn and of energy prices (substitution effects induced by coal and gas prices), which are estimated at around 300 and 200 million tonnes, respectively. Unfortunately, the potential impact of EUA carbon prices on emissions cannot be identified mainly because of the insufficient number of data points available.

Berghmans *et al.* (2014) assess the relative importance of different factors, including electricity production from RESs, economic activity, energy prices and technical plant features, in explaining CO_2 emissions from the European electricity sector, over the first two trading periods of the EU ETS (2005-2012). The authors fit econometric panel-data models for installation-level annual emissions, using the EU Transaction Log[76] and the World Electric Power Plants database as primary information sources. The results show that, in the electricity sector, emission abatement came to a large extent from RES deployment displacing electricity generation from fossil fuel plants. While significant effects are found also for economic activity as well as for energy and carbon prices, the roles of these were clearly secondary as compared to that of RES expansion.

5.2.2 Abatement cost of complementary policies

Marcantonini and Ellerman (2016) study the cost of reducing CO_2 emissions through RES generation, as measured by what they call "the implicit carbon price". Focusing on Germany, the authors estimate the implicit carbon prices of wind and PV electricity generation, 2006-2010. The implicit carbon price is given by the ratio of net costs to emission reductions, both as related to RES generation. With reference to net costs (the numerator), the remuneration to RES generation, which crucially depends on the RES incentives paid by electricity consumers[77], are the main cost component; while fuel cost

[76] The EUTL records official information both on regulated installations and allowance transactions in the EU ETS.

[77] In Germany, as in the vast majority of European countries, the costs of support to RES generation are recovered through specific surcharges on final electricity prices. As electricity is a necessary consumption good (the income elasticity of electricity demand is indeed typically very low), the

savings, achieved by the displacement of conventional generation, are the main benefit component. Both the fuel cost savings and emission reductions are derived through simulation, using a unit commitment model of the national electricity system. For wind electricity generation, the implicit carbon price is found to have been in the order of tens of euros per tonne of CO_2, averaging €57/tCO_2 over 2006-2010. For PV generation, the implicit carbon price is much higher, in the order of hundreds of euros per tonne of CO_2, averaging €552/tCO_2 over the same five-year period. Using the same approach, Marcantonini and Valero (2017) estimate the implicit carbon prices again for wind and PV electricity generation, this time in Italy, 2008-2011. As in Marcantonini and Ellerman (2016), reducing emissions through wind electricity generation turns out to be much cheaper than doing it by producing PV electricity. For both technologies, however, implicit carbon prices are much higher in Italy than in Germany, averaging €165/tCO_2 (wind) and €1000/tCO_2 (PV), over this period. The implicit carbon price depends on several factors, but primarily on the level of financial incentives given to RES generation and on the type of conventional generation displaced by RES injections.

As part of a wider investigation into the interactions among climate and energy policies in the EU, Rey *et al.* (2013) estimated the cost of CO_2 abatement for seven RES-E technologies, namely hydro, wind, biomass, biogas, PV, geothermal and waste, in eight European countries (the Czech Republic, France, Germany, Italy, the Netherlands, Poland, Spain and the UK). The analysis is carried out only for 2010 and, more important, the approach is simpler than the one used by Marcantonini and Ellerman (2016) (or Marcantonini and Valero, 2017). The abatement cost of a technology is given by the ratio of *a)* the average financial support to the technology

distributional incidence of these surcharges is markedly regressive. This has raised equity concerns (see, e.g., Grösche and Schröder, 2014; Verde and Pazienza, 2016).

(€/MWh)[78] to *b)* the corresponding amount of avoided CO_2 emissions. Benefits in the form of fuel cost savings are not considered and emission reductions are estimated by applying average values of electricity CO_2 intensity[79]. The results vary widely across technologies and countries, reflecting different levels of financial support and different types of energy mix. PV exhibits by far the highest abatement costs, with values over €700/tCO_2 in most countries. As to the other technologies, abatement costs always exceed €50/tCO_2 (with the notable exception of wind, in the Czech Republic) and in most cases fall in the range between €100 and €200/tCO_2.

5.2.3 Effect of complementary policies on carbon prices

Using time series analysis and EU-level monthly data, Koch *et al.* (2014) investigate the relative importance of *a)* relative prices of natural gas and coal, *b)* expectations on economic activity (as measured by the STOXX EUROPE 600 stock index return or, alternatively, by the change of the Economic Sentiment Indicator), *c)* growth of RES-E generation (hydro, wind and PV) and *d)* use of international offsets (proxied by the number of Certified Emission Reductions[80] issued), in explaining the dynamics of EUA carbon prices. They analyse the period from January 2008 to October 2013, a time interval during which carbon prices fell from almost €30, in mid-2008, to less than €5, in mid-2013. The first main finding is that only about 10% of the variation in EUA

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[78] Financial support is the weighted average cost per MWh of the annual expenditure on RES-E support schemes as estimated by CEER (2013).

[79] To calculate avoided emissions, alternative average values of electricity CO_2 intensity are taken into account. Namely, those referring to: *a)* the country's electricity system (as if RES electricity only displaced national generation); *b)* the European electricity system (as if RES electricity displaced generation at the EU level); and *c)* natural gas generation (as if RES electricity only displaced generation from natural gas).

[80] Certified Emission Reductions (CERs), which are issued under Kyoto Protocol's Clean Development Mechanism (CDM), can be used for compliance in the EU ETS (though not as of Phase IV, i.e. after 2020). The use of CERs for compliance affects EUA prices because it implies a reduction in the demand for EUAs.

prices can be explained by the four variables of interest, despite their being abatement-related “fundamentals” of carbon prices (Hintermann *et al.*, 2016). Secondly, within this 10%, 40% of the variation in carbon prices can be attributed to the variation in expected economic conditions and almost 25% to RES-E generation, thus pointing to moderate interaction effects with overlapping RES policies. While being statistically significant, both fuel switching and (especially) international offsets have very little explanatory power. The authors conclude that, as it turns out, EUA price dynamics cannot be solely explained by marginal abatement cost theory. Structural weaknesses – and a lack of credibility in particular – may be at the root of the inefficient carbon pricing mechanism.

6. Conclusions

The specific purpose of an ETS, such as the EU ETS, is to minimise the total cost of achieving the given emission reduction target. In principle, any additional policy that directly or indirectly contributes to reducing emissions from the sectors covered by the system would increase this cost. Reality is more complex than this, however. The adoption of policies (or instruments or measures) additional to an ETS, in the sense specified, can be well-justified. This is true for the case of market failures, which justify, for example, support schemes for R&D activities and – though here there may be less of a consensus – for the deployment of certain technologies, as with several RES policies. These are policies responding to market failures that an ETS alone would not address and which would benefit its dynamic efficiency. Moreover, additional policies may be primarily motivated by objectives that are different from climate mitigation. For example, this is the case of EE support policies adopted for enhancing energy security. As Sorrell and Sijm (2003) put it, with reference to emissions trading in the policy mix, “there will be trade-offs between long-term and/or non-efficiency objectives and short-term increases in abatement costs.”

The theoretical or ex-ante literature on the interactions between the EU ETS, or a hypothetical ETS, and other policy instruments is substantial. But the empirical literature providing relevant ex-post evaluations for such interactions is rather small. Within this, the literature extending beyond the electricity sector is particularly scant. Each and every one of the empirical studies that we found focuses on one of the following general questions: *a)* How much abatement was due to complementary policies (notably, RES-E policies) rather than to the EU ETS?; *b)* What was the abatement cost of RES-E policies?; and *c)* What was the impact of complementary policies on EUA carbon prices? Concerning the first research question, the findings of different studies are remarkably consistent with each other. They show that much greater proportions of emission reductions (relative to BAU scenarios) were due to RES policies rather than to the EU ETS. Of course, differences in the amount of abatement vary depending, in the first place, on the magnitude of RES investment in the given country and on whether only the electricity sector or all regulated sectors are considered. In any case – moving to the second research question – effectiveness in reducing emissions does not imply cost-effectiveness. Here as well, the relevant studies agree in showing that abating emissions through RES-E deployment was generally expensive and, most probably, costlier than would have been the case using the EU ETS alone. Yet, crucially, RES-E policies can be interpreted as serving multiple objectives, in which case, a simple comparison of their abatement cost with the level of carbon prices would be unfair. Besides, they are intended to address market failures. They, thus, provide economic benefits, which, however, are very difficult to quantify. In this connection, the contribution of only one study assessing the impacts of RES support policies on GDP and on total employment is noteworthy: the results are generally positive, duly taking into account the positive macro-economic effects of related investments. As to the third research question, only one study has taken it up. Koch *et al.* (2014) find that current values of the fundamentals related to marginal abatement cost theory, among which RES generation is included, explain only a small proportion of the variations in EUA carbon prices. An implication of this result seems to be that expectations on the future course of the European carbon market are even more central in price formation than one might expect. Further investigation is warranted, however.

The growing efforts of the EU to implement a mix of instruments that efficiently serves the objectives of climate policy and energy policy are clearly visible. It is enough to think of the 2020 Climate and Energy Package, the 2030 Climate and Energy Framework, the Energy Union, and the Clean Energy Package. With specific reference to the EU ETS, the recently established MSR directly tackles the most serious side effect that other policies and, for that matter any relevant external factor, may have on the dynamic efficiency of the European carbon market. By introducing some flexibility into the supply of emission allowances, the MSR is expected to shelter the EU ETS, to some extent, from negative demand shocks and, thus, ultimately, to prevent overly low carbon prices. Different views exist as to whether the MSR will satisfactorily achieve its purpose and whether it is the best possible approach. For the time being, we note that the MSR's combination of a rule-based mechanism and the periodic revision of its parameters (every five year) seems a reasonable solution considering the ever-changing environment external to the EU ETS.

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Chapter 4

Low-carbon innovation and investment

1. Introduction

As time is ticking away for drastically reducing global greenhouse gas (GHG) emissions, sustained innovation and diffusion of low-carbon technologies become increasingly important for controlling the cost of this endeavour. The EU, which is committed to achieve a climate-neutral economy by 2050 (European Commission, 2018), has been on the front line and wants to continue leading by example. The stated objective of the EU ETS, which is the core piece of the EU's strategy for decarbonising the economy, is "to promote reductions of greenhouse gas emissions in a cost-effective and economically efficient manner" (European Parliament and Council, 2009). Cost effectiveness is to be achieved both in the short term and long term. In slightly more technical jargon, this means that the EU ETS aims to be both "static-efficient" and "dynamic-efficient". However, while static efficiency is almost a natural feature of cap-and-trade schemes like the EU ETS (net of possible inefficiencies of the allowance market), dynamic efficiency cannot be taken for granted. This is because carbon prices are only one of many variables that subject operators consider in making climate-related investment decisions. Nevertheless, though possibly not a sufficient condition, significant carbon price levels are necessary for stimulating low-carbon innovation and investment. The regulation of the interplay between the supply and demand of emission allowances is thus central. So is any other policy lever that can influence operators in making climate-related investment decisions.

Following the economic crisis, the persistent imbalance between the supply and demand for allowances, and the corresponding impact on the carbon price signal, have led to concerns as to whether existing incentives are sufficient for spurring the necessary levels of innovation and diffusion of low-carbon technologies. Indeed, most analysts would agree that this has become the main problem of the EU ETS. Steps have been taken to address the surplus of allowances and, thereby, this weakness of the scheme in terms of dynamic-efficiency. Notably, the recent reform for Phase IV (2021-2030) both tightens the supply of emission allowances and significantly enhances the existing programme of financial support for low-carbon innovation and investment. Moreover, not long before the reform for Phase IV, the surplus of allowances was first tackled through the "backloading" of 900 million allowances and, then, in a structural way, with the establishment of the Market Stability Reserve (MSR).[81]

In view of the above, empirical analyses of the impact of the EU ETS on low-carbon innovation and investment are precious for assessing the more general performance of the policy instrument. However, the picture offered by the existing empirical literature is less comprehensive and especially less up-to-date than ideal. Econometric studies, in particular, are few due to the lack of suitable and accessible databases. This lack of information (statistics) is a problem that should be addressed if empirical analysis is to inform policy. Moreover, the theoretical underpinnings concerning the process of technological change are not always clearly recognizable in the empirical investigations. For example, sometimes there seems to be confusion between concepts such as "invention", "innovation" and "diffusion". That said, some

[81] The MSR is a rule-based mechanism that limits the excess supply of emission allowances. It is operative since January 2019. The MSR is discussed in Chapter 3.

robust conclusions can be drawn from the empirical literature, as we show in this chapter.

The rest of the chapter is organised as follows. Section 2 recalls some fundamentals of environmental innovation. Section 3 focuses on the concepts of static and dynamic efficiency with reference to the EU ETS. Section 4 illustrates the EU ETS funding programmes for low-carbon innovation. Section 5 summarises the relevant empirical evidence. Section 6 concludes.

2. Basics of environmental innovation

Over the past three decades, interactions among environmental policy, technological change and the environment have become an increasingly relevant topic in the dialogue between economists and policymakers. There are at least two reasons for this. First, the impact of economic activity on the environment depends critically on the pace and direction of technological change. As far as climate change is concerned, green technologies allow for a reduction in GHG emissions by replacing exhaustible resources or by enabling their more efficient use. Second, induced technological change can affect the balance between the costs and benefits of environmental policy, with implications for welfare outcomes.

In Schumpeter's (1942) theoretical framework, technological change is sectioned into three distinct phases: *a)* "invention", which coincides with the development of a new product or process; *b)* "innovation", which is accomplished only when the new product or process enters the market; and *c)* "diffusion", which occurs when a successful innovation is progressively adopted by other firms in the market. Importantly, not all inventions necessarily evolve into innovation. Moreover, in Schumpeter's view, while a requisite for innovation is extra profit or a temporary monopolistic position, inventing something is not strictly necessary. Innovation without invention is, in principle, possible. This is the case with organisational innovation. Now focusing on environmental innovation (or eco-innovation), a most consolidated definition describes it as "the production, application or use of a product, service, production process or management system new to the firm adopting or developing it, and which implies a reduction in environmental impact and resource use (including energy) throughout its life cycle" (Kemp, 2010). An important implication of this definition is that environmental innovation does not need to be "new to the market": being "new to the firm" suffices for it to be considered as such (Kemp and Pontoglio, 2011).

In analysing the interaction between environmental policies and technological change, it is important to distinguish between innovation and diffusion. Let us focus on innovation, first. Here, the earliest relevant contribution dates back to the work by Hicks (1932), which stresses how an increase in the relative price of a production factor spurs inventions aimed at reducing its use. This basic idea underlies what is known in economics as "induced innovation hypothesis" (IIH). Its application to the realm of environmental policy is straightforward. As environmental policies generally imply (directly or indirectly) an increase in the price of one or more environment-related production factors, the IIH suggests that firms will respond to regulation with innovation allowing them to reduce their use of those. Empirically testing for such policy-induced innovation effects is not straightforward, however. The main challenge in testing for the policy-inducement mechanism is that, ideally, researchers would study the relationship between the shadow price of pollution (i.e. the implicit price of abating pollution) and innovation; yet, such shadow prices cannot be observed (Popp *et al.*, 2010). To overcome this difficulty, the empirical literature uses a range of strategies to proxy for environmental policies. [82]

[82] For example, Gray and Shadbegian (1998) use expert judgement about relative regulatory stringency; Demirel and Kesidou (2011) construct a set of binary variables reflecting the existence of environmental

A different perspective on the relationship between environmental policies and innovation is found in the Porter hypothesis (PH) (Porter and van der Linde, 1995). The PH stems from Simon's (1947) idea of bounded rationality, whereby firms engage in satisfying rather than in maximising behaviour. Accordingly, in the absence of external shocks, "satisfying" firms may not have an incentive to search for all possible opportunities to increase profits if things are going reasonably well. Under this paradigm, environmental policies, by imposing external constraints onto firms, can actually push these to search for new profit opportunities. Satisfying firms may not respond to regulation only by introducing input-saving technological change, but they may also be induced to search for and discover new alternative production processes. There are at least five ways in which this mechanism may operate, which correspond to the following functions of environmental regulation: *a)* signalling possible resource inefficiencies (internal to the firm); *b)* raising corporate awareness; *c)* reducing uncertainty in pollution reduction investments; *d)* motivating cost-saving innovations (by putting pressure on firms' production costs); and *e)* contrasting free riding behaviour (as firms cannot opportunistically gain on their rivals by avoiding environmental investments). Concerning the ensuing economic effects, the literature usually distinguishes between two versions of the PH: the strong version posits that productivity and competitiveness gains induced by regulation more than offset compliance costs; the weak version states only that environmental regulation stimulates environmental innovation.[83] The empirical literature is not conclusive as to whether evidence more frequently supports the strong version of the PH (Brännlund and

standards or environmental taxes; Lanjouw and Mody (1996) uses pollution abatement and control expenditures (PACE) on the part of firms.

[83] The third "narrow" version of the PH argues that only certain types of environmental regulations can foster competitiveness (Jaffe and Palmer, 1997).

Lundgren, 2009; Ambec *et al.*, 2013; Barbieri *et al.*, 2016; Dechezleprêtre and Sato, 2017).

Let us now turn to the diffusion phase of technological change. Standard innovation theory defines diffusion as the process through which a new technology penetrates the relevant market and becomes available to most firms (Popp, 2010a). While innovation can be disruptive, diffusion is a gradual, possibly a slow process. Seminal theoretical contributions in this field (e.g. Geroski, 2000) show that the share of adopters of a new technology in a market draws an "S" over time; that is, few early technology adopters characterise the first stage of diffusion, which is followed by a period of rapid mass-adoption and, finally, slow adoption when the market reaches maturity. Empirical studies show how firm's characteristics, as opposed to targeted policies, can explain much of the variability in adoption rates. However, as many studies also show, this general result does not really apply to the adoption of environmental technologies. Pollution reducing end-of-pipe technologies, for example, are most unlikely to be adopted by firms in the absence of environmental regulation setting binding constraints. In this connection, the concept of "induced diffusion" indicates the adoption of existing technologies driven by environmental regulation.

3. The static vs dynamic efficiency of emissions trading

There is a fundamental short-term/long-term distinction when assessing the efficiency (i.e. the cost effectiveness) of policies for climate-change mitigation. Over the short term, cost minimization depends on agents' operational decisions, which are conditional on existing capital and technology. In the long term, cost minimization also depends on investment decisions, which determine capital and technology through time. Accordingly, the EU ETS can be assessed both with respect to its static (short-term) efficiency and with respect to its dynamic (long-term) efficiency.

In principle, cap-and-trade schemes such as the EU ETS are static-efficient by nature: emissions are cut at the least total cost, as the market mechanism equalises marginal abatement costs across regulated entities[84]. By contrast, establishing whether the EU ETS is dynamic-efficient, that is, whether it induces investments that allow minimizing abatement costs over the long term, proves more complicated. Operators' expectations on investment returns, and, thus, their investment decisions, are affected by uncertainties intrinsic to the EU ETS. These concern future carbon prices and possible regulatory changes, as well as a multitude of other market and policy factors. Moreover, while in principle the EU ETS is static-efficient, independent of the way the allowances are distributed[85], the same is not true for its dynamic efficiency. For example, ex-ante, whether allowances are received for free or not may change the relative convenience of alternative investments and, therefore, investment decisions.

4. The EU ETS funding programmes for low-carbon innovation

Irrespective of its level, the role of carbon pricing in spurring low-carbon innovation is not exhaustive. Above all, since innovative projects are risky by nature, they require specific forms of financing.[86] NER 300 is the EU ETS funding programme for highly innovative low-carbon projects in the pre-commercial demonstration phase.[87] The sale

[84] According to the independence property, the market equilibrium in a cap-and-trade system is cost-effective and independent of the initial allowance allocation. Hahn and Stavins (2011) identify a number of factors that can lead to the independence property being violated, including transaction costs, market power and price uncertainty.

[85] Operational decisions are independent of the way allowances are acquired. In either case, using allowances for compliance entails a cost: a real cost, with auctioning; an opportunity cost, with free allocation.

[86] For a discussion of the rationale for and recommendable features of public support for low-carbon innovation, see Mazzucato and Semieniuk (2017) and Nemet et al. (2018).

[87] NER300 is established by Art.10a(8) of the EU ETS Directive. The rules of the funding scheme are specified in Commission Decision 2010/670/EU.

of 300 million emission allowances in the New Entrants Reserve of the EU ETS[88] generated €2.1 billion, which were awarded in two rounds (in December 2012 and July 2014) to one CCS project and to 38 renewable energy projects. As only three of the 39 projects have so far entered into operation, it is too early to draw definitive conclusions on the programme's performance. Nevertheless, some lessons have already been learnt.[89] There are three critical points limiting NER 300's effectiveness. First, as a rule, funds are given out upon a project's success.[90] While this is an attempt to guarantee the good use of public money, it fails to take into account that innovation is, by its nature, risky. Second, funds can only cover up to 50% of the project's additional innovation costs. This condition is to make firms leverage additional funding from other sources. But the project promoter may struggle to secure the extra financing. Third, available funding may be insufficient for the more expensive projects involving breakthrough technologies such as CCS. [91]

The proposed revision for Phase IV introduces the Innovation Fund (IF), which will be an enhanced version of NER 300. At the time of writing (December 2018), only a few basic features of the IF have been defined. The scope of the programme will be wider compared to that of NER 300, both in terms of the resources made available and in terms of the range of eligible beneficiaries. The financial resources will be generated by the sale of up to 450 million allowances and industrial sector projects will be funded, too. The maximum funding rate will be raised to at least 60% and fund disbursement depends, in part, on the achievement of milestones in the project's development. Recent

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[88] The New Entrants Reserve is an EU-wide pool of emission allowances set aside for new and expanding installations eligible for free allocation.

[89] The impact assessment accompanying the proposed revision for Phase IV (European Commission, 2015) provides a detailed account of NER 300 and useful information on the future Innovation Fund.

[90] Up to 60% of the awarded funding can be provided upfront, but conditionally on a MS providing an appropriate guarantee.

[91] The funding awarded to the one CCS project (€300 million) only covers 34% of its additional costs.

consultations that the EC has conducted with the stakeholders suggest that further changes will be adopted. [92] While the IF will still use grants, these will be accompanied by other ways of financing eligible projects. Some diversification in financial instruments, including loans, guarantees or equity, may be tailored to fit projects with different risk profiles. This would result in more financial resources being leveraged. Furthermore, the project's ranking for selection may not be exclusively based on individual "value for money" in terms of emissions abatement.[93] The potential for wider benefits emerging from cross-sector cooperation may also be taken into consideration.

5. The empirical evidence

There are several but not plentiful empirical studies analysing the impact of the EU ETS on low-carbon innovation and investment.[94] Above all, the relevant literature as a whole falls short of providing exhaustive evidence. Crucially, the evidence we have is limited to Phases I (2005-2007) and II (2008-2012). This means that possible effects of the changes to allowance allocation introduced with Phase III on innovation have not yet been appreciated: particularly worthy of note here is auctioning and the application of emission efficiency benchmarks.

Our review of the literature resulted in 21 works being identified. They are here divided into two groups and presented accordingly. There are the studies using econometrics, while the others provide qualitative and descriptive analyses. There are only six econometric studies owing to the paucity of suitable databases – which seems to be a real limiting factor. But, naturally, their results have some defined statistical

[92] The results of a series of workshops instrumental in the design of the IF are summarised in Climate Strategy (2017).

[93] Under NER 300, projects were ranked by their cost-per-unit performance, the performance being the amount of renewable energy produced or the amount of CO_2 stored in the case of CCS projects.

[94] Studies looking at the effects of the EU ETS on the competitiveness of regulated firms are far more numerous, for example.

relevance. Conversely, the non-econometric studies are significantly more numerous (15), but their external validity is limited or undetermined. The non-econometric studies generally draw on less empirical information, which is often generated by the authors themselves through ad-hoc interviews. They are a useful complement to the econometric literature given the insights that they offer.

Our accounts of the two literature subsets are presented in different ways. While the econometric studies are summarised one by one, the non-econometric ones are combined into a single narrative. In both cases, general conclusions are derived.

5.1 Econometric literature

Calel and Dechezleprêtre (2016) estimate the impact of the EU ETS on low-carbon patenting. It is a study that stands out in the literature for four reasons. First, patent counts are an objective measure of innovation (invention); albeit only a proxy for the number of innovations that translate into new production processes or products. Second, the firms in the sample operate over 80% of all EU ETS installations. Third, the approach used, namely difference-in-differences (DiD), offers a clear-cut causal interpretation of the estimated effect. Fourth, the main result for the magnitude of low-carbon innovation brought about by the EU ETS is significantly more positive than most other studies would suggest. Indeed, the authors estimate that the EU ETS caused an increase of up to 36%, over 2005-2009, in the number of low-carbon patents granted to regulated firms. Other relevant results from the study are: *a)* that no evidence of an indirect innovation effect on non-regulated entities is found; and *b)* that the surge in the total number of low-carbon patents observed over the period in question was primarily driven by rising energy prices.

Martin *et al.* (2013) is a second important econometric study investigating the determinants of low-carbon innovation, here measured by any R&D activity aimed at curbing emissions or energy consumption or at developing products that can help customers to reduce their emissions. It uses primary data from interviews conducted in 2009 with the managers of 770 manufacturing firms in six European countries. The study finds several interesting results. First, the responses indicate that most firms are engaged in climate-related innovation, and that this effort was focused on process innovation rather than product innovation. Crucially, however, no statistically significant difference is found between firms regulated by the EU ETS and non-regulated ones. Second, significant differences in the propensity to innovate were found across sectors and countries. Third, low-carbon innovation is positively associated with firms' expectations about the stringency of their future allocation. Fourth, firms positioned just below the established thresholds for receiving free allowances in Phase III[95] engaged more strongly in low-carbon innovation. This last result is presented as evidence of free allocation having a causal negative effect on low-carbon innovation.

Schmidt *et al.* (2012) analyse the effects of the EU ETS and long-term emission reduction targets, as they are perceived (rather than as they are according to certain objective metrics), on investments in technology adoption and on R&D. The study is based on interviews with managers of 65 electricity-producing firms and 136 firms providing technology for electricity generation, in seven European countries. The effects are measured by the respondents' answers to the questions about the intensity and the direction of innovation investments between two periods: 2000-2004 and 2005-2009. Among other results, the authors find evidence that not only did the grandfathering of emission allowances hamper low-carbon technology adoption it effectively incentivised the adoption of emitting technologies. Long-term emission reduction targets are, nonetheless, an important R&D trigger.

Focusing on Sweden and using the DiD approach (without pre-treatment matching), Lofgren *et al.* (2014) test for whether participation in the EU ETS stimulated

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[95] Such thresholds are based on sectoral trade and carbon intensities. For details, see Chapter 1 which illustrates the rules governing allowance allocation through the trading periods.

investment in low-carbon technologies. The estimation sample comprises some 706 Swedish firms, of which 229 are in the EU ETS, from all regulated sectors, plus five control sectors not covered by the scheme: the study covers the years 2000 to 2008. Conducting separate but similar analyses for investment below and over €1 million (in a year), different logit model specifications are estimated, with the outcome of interest being the probability of low-carbon investment. The main result is that statistically significant effects of the EU ETS are not found, either for small or for large investments. [96] Though limited to Swedish firms, the study also reports some insightful descriptive statistics on low-carbon investment within and outside the EU ETS sector. [97]

Bel and Joseph (2015) investigate whether allowance oversupply had a negative effect on the number of patents for climate change mitigation technologies filed at the European Patent Office. Data for 28 countries are used, covering the period 2005-2011. Some interesting descriptive statistics are first reported on the country distribution of climate-related patents, showing Germany to be the clear innovation leader in Europe. Country-level panel data models are then fitted for annual patent counts, explained by allowance oversupply – the variable of interest – and a number of sectoral and economy-wide control variables: GDP, sectoral activity indices, R&D spending, number of employees in the service sector, share of renewable energy, among others. In all model specifications, allowance oversupply is found to be negatively correlated with the number of climate-related patents. The result is, thus, consistent with the weak

[96] Considering the frequency of investment decisions, the analysis would have probably benefited in a significant way from longer time coverage.

[97] For example, over the period 2002-2008, out of 229 regulated firms, 70 (i.e., almost a third) made some low-carbon investment, 40 of which invested more than €1 million in a given year. With reference to the same firms, 46% of the large investments were in biofuels, 25% in district heating and 22% in energy efficiency; by contrast, over half of the small investments were in energy efficiency. The pulp and paper industry and the energy and heating sector had the highest shares of investors. Moreover, for regulated firms, fuel use was higher among investors compared to non-investors; the converse is true for non-regulated firms.

version (at the least) of the Porter Hypothesis, whereby environmental innovation is stimulated by more stringent regulation. The authors intend to extend the analysis in future work, in which case the first desirable variation would be the use of firm-level data.

Borghesi *et al.* (2015a) investigate whether the EU ETS affected the likelihood of climate-related investment (distinguishing between energy efficiency and emission reduction as aims) undertaken by Italian manufacturing firms, 2006-2008. Firm-level probit models are estimated, using Eurostat's Community Innovation Survey as the primary data source. Several explanatory variables are considered: some internal to the firm, some external to the firm (still, firm-specific) and policy factors. The effects of the EU ETS are specifically captured: *a)* by a dummy variable identifying regulated sectors; and *b)* (in regressions limited to observations of regulated sectors) by a stringency indicator given by the ratio of sectoral emissions to allocated allowances.[98] As expected, firms in regulated sectors are more likely to make climate-related investments.[99] The allocation stringency indicator, however, turns out to be negatively correlated with innovation. The authors suggest that this result may be related to one or more of the following: firms' anticipatory behaviour; sectoral idiosyncratic factors and reverse causality between allocation and innovation. Yet, further investigation is not undertaken due to data limitation.

All in all, while the results vary, to some extent, across sectors and countries, robust evidence emerges from the above econometric literature. First, the role of the EU ETS in driving low-carbon innovation is generally judged to have been limited, especially when compared to that of energy prices. In Phases I and II, the EU ETS

[98] Four regulated manufacturing sectors are analysed: coke and refining; ceramic and cement; paper and cardboard; and metallurgy.

[99] In the study, firms' participation in the EU ETS is defined at the sectoral level. This implies some measurement error, since not all firms in a regulated sector fall under the EU ETS.

appears to have fostered incremental emission reductions realised by regulated firms through fuel switching and small-scale energy efficiency investments. Seemingly discordant with this view are the findings of Calel and Dechezleprêtre (2016), which show the innovation effect of the EU ETS, as measured by patenting, to have been substantial. A legitimate hypothesis for explaining this discrepancy is that a significant share of newly patented low-carbon innovations did not translate into operating production processes or commercialised products. Second, there is a clear consensus that the surplus of allowances in the carbon market and the corresponding impact on the carbon price negatively affected low-carbon innovation and investment. Third, allowance grandfathering – a defining feature of the EU ETS in Phases I and II – is also identified as having been a major factor in limiting the ability of the EU ETS to incentivise low-carbon innovation and its uptake.

5.2 Non-econometric literature

Most of the studies on the impact of the EU ETS on low-carbon innovation and investment provide qualitative or descriptive analyses. We found fifteen such studies, most of which (twelve) are based on primary information collected through ad-hoc interviews, typically with managers of regulated firms. These studies tend to be country and sector specific. Five of them focus on the electricity sector, five on the paper sector and one on the chemicals sector. The remaining four cover multiple sectors. Some of them are sufficiently narrow in scope to qualify as case studies.

Among the insights offered by this literature, those regarding the (intended or perverse) incentives of regulation affecting firms' low-carbon innovation and investment are most relevant. Indeed, while the econometric literature deals with the outcomes of such incentives, it does not focus on their workings. Further specific insights of the non-econometric literature concern organisational innovation, which is a type of process innovation. According to Rogge's (2016) taxonomy, organisational innovations can relate to firms' business practices (e.g., implementing monitoring, the verification and reporting of emissions, or participating in carbon trading), workplace organisation (e.g., making the EU ETS a top management issue) or external relations (e.g., collaboration with other firms or public institutions).

Below, we first discuss the main evidence from the studies that exclusively look at the electricity sector. We, then, turn to those analysing other or multiple sectors.

5.2.1 Electricity sector

The dynamics of low-carbon innovation and investment in the electricity sector has been of special interest to researchers. There are at least two reasons for this. First, the electricity sector is expected to be fully decarbonised before any other. Second, the electricity sector underwent a major change in its allowance allocation regime in passing, with Phase III, from grandfathering to full auctioning.[100] The allocation regime matters in the case of the EU ETS, as certain rules – the new-entrant and closure provisions – can influence long-term decisions on the type (or location) of a new plant and on the extended use of an old one. The perverse incentives associated with the new-entrant and closure provisions, for, respectively, new and closing installations are frequently stressed.[101] Granting allowances to new installations improved the economic attractiveness of investments in CO_2-intensive electricity generation (Rogge *et al.*, 2011). Closure provisions, whereby operators forfeit the allowances allocated to the installations they close, incentivised the protracted operation of high-emitting installations (through, e.g., retrofits). These perverse incentives vanished with Phase III, as electricity producers (with the exception of those in some of the new member

[100] Other sectors will undergo a similar (though more progressive) transition in the future, so understanding this precedent is useful in view of the coming change.

[101] New-entrant and closure provisions characterise the EU ETS. Previous cap-and-trade schemes did not foresee such provisions (see, e.g., Ellerman et al., 2010).

states) ceased to receive free allowances. In fact, even before the start of Phase III, there is evidence that the very expectation of auctioning caused the cancellation of planned investments in coal-fired power plants (Hervé-Mignucci, 2011).

The same literature documents the initial interest in R&D activities related to Carbon Capture and Storage (CCS). It was, however, an interest that did not last long due to both the high cost of CCS, given the level of carbon prices (too low to make such investments economically viable), and issues of local social acceptance (Hoffman, 2007; Rogge and Hoffmann, 2010; Rogge *et al.*, 2011). Indeed, the expectation of sufficiently high carbon prices in the future and regulatory predictability are the elements that seem to matter most for RD&D (research, development and demonstration) decisions. Overall, the specific impact of the EU ETS on low-carbon innovation and investment in the electricity sector was moderate. Most often, what impact there was came in the form of small-scale investments with short amortisation times (e.g., three to five years) improving the efficiency of existing conventional plants. Table A1, in the Appendix, summarises the main features and the main results of this literature.

5.2.2 Other sectors

Considering the large number of sectors that fall under the EU ETS and the limited size of the literature at hand (summarised in Table A2, in the Appendix), the pulp and paper sector has received a surprising amount of attention. The five non-econometric studies that specifically look at this sector are concordant in the conclusion that the EU ETS had only a very small effect, if any, in driving low-carbon innovation and investment. Carbon prices were too low and allowance allocations too abundant to affect investment decisions. A second related common conclusion is that market factors, notably the cost of raw material and especially energy costs, were far more important in explaining investment decisions. The impact of the EU ETS is, indeed, mainly felt indirectly through energy costs (Gulbrandsen and Stenqvist (2013). According to Pontoglio (2008), the relevant investments mainly consisted in the adoption of existing technologies for improving core process efficiency. The limited effect that the EU ETS had on innovation in the pulp and paper sector is also confirmed by Fontini and Pavan (2014). Performing an index decomposition of the sectoral emission trends in Italy, the authors show that technological change was never a relevant contributor to emission reductions. For Phase I, a compositional effect is found, whereby emissions slightly declined thanks to a shift toward less polluting types of paper products. For the second trading period, the reduction in emissions was due to lower output – the effect of the economic crisis.

The remaining non-econometric studies bring additional evidence that is mostly consistent with the results already discussed. Analysing the chemicals sector in Portugal, Tomas *et al*. (2010) find that the EU ETS had no relevant effects on production costs and that grandfathering allocation failed to incentivise innovation activities. Similarly, though with reference to the ceramics, lime and cement sectors, in Belgium, Venmans (2016) concludes that uncertainty and free allocation were the main factors hindering low-carbon innovation. Based on interviews with representatives of different industries in eight European countries, Borghesi *et al*. (2015b) investigate whether and how policies and market factors affected eco-innovation. The EU ETS turns out to have had a relatively more significant role in the electricity sector than in manufacturing sectors, spurring fuel switching and CCS-related activities in the former. Based on questionnaires administered to regulated firms in Ireland, Anderson *et al.* (2011) find that significant low-carbon innovation and investment took place during Phase I. Substantial proportions of the respondents answered that they had undertaken some technological changes, fuel switching, and behavioural change. However, it is unclear to what extent these were specific effects of the EU ETS. Finally, Petsonk and Cozijnsen (2007) document a few cases of firms in the agriculture sector benefitting from different process innovations brought about by the EU ETS. All in all, this subset of the non-econometric literature reinforces the view that the impact of the EU ETS on low-carbon innovation and investment was moderate at best. Other factors, energy prices above all, tended to be more relevant variables. On the other hand, robust evidence of organisational innovation related to the EU ETS suggests that the policy instrument has been internalised by most regulated firms.

6. Conclusions

The success of the EU ETS is vital for the full decarbonisation of the European economy. In turn, continued progress in low-carbon innovation and investment on the part of firms regulated by the EU ETS is necessary for meeting the EU's long-term mitigation targets and for minimising related costs. Cost effectiveness in emissions abatement, both in the short term and in the long term, is the very purpose of emissions trading. In this sense, however, the protracted low level of EUA prices has been threatening the ability of the EU ETS to fulfil its own mission. The level of carbon prices (both present and future) is, indeed, the primary parameter determining the strength of the incentives for low-carbon innovation and investment.

The empirical literature on the effects of the EU ETS on low-carbon innovation and investment is not huge and, above all, fails to present a full picture of the current state of things. To date, not a single ex-post analysis exists that comes up to the beginning of Phase III. This is both surprising and regrettable because major regulatory changes, likely to benefit the scheme's dynamic efficiency, were introduced with the third trading period: notably, the switch from grandfathering to auctioning in the electricity sector and the application of emission efficiency benchmarks. Moreover, within the existing empirical literature, the number of econometric studies is particularly small. While both econometric and non-econometric (descriptive, qualitative, case study) analyses are needed, as they can offer complementary evidence, the relevance of the results in terms of statistical significance is only clear for the former. The lack of suitable (and accessible) databases is the main reason for the scarcity of econometric contributions in this field. More firm-level data on low-carbon investment (ideally, both for regulated and non-regulated firms, so as to have more possibilities for causal analysis) are badly needed.

Nevertheless, some robust conclusions can be drawn based on the literature on Phases I and II:

A) While the EU ETS alone did not stimulate major low-carbon investments, it did stimulate some small-scale investments typically described as small-scale with short amortization times (e.g., three to five years), resulting in incremental emission reductions.

B) One prominent econometric study (Calel and Dechezleprêtre, 2016) finds that the EU ETS brought about a substantial increase in the number of low-carbon patents granted to regulated firms. This result is comforting, but we do not know how many (or what proportion) of these additional patented innovations turned into new processes or products.[102] Here, using Schumpeter's taxonomy of technological change, it would be useful to gather data (currently lacking) and, thus, evidence not only on the transition from invention to innovation, but also from innovation to diffusion.

C) In Europe, the observed surge in the total number of low-carbon patenting was primarily driven by rising energy prices. Similarly, energy prices were much more important determinants of decisions on low-carbon investment than carbon prices.

D) Heterogeneity in the propensity to innovate is significant across sectors and countries. However, evidence is scattered and again, apart from patent data, there is not anything like a comprehensive mapping of low-carbon investments across Europe.

[102] Conversely, there may be new production processes or products that are not patented.

E) Low-carbon innovation efforts have focused on production processes much more than on products.

F) Free allocation hampered low-carbon investments. One (perfectly rational) explanation relates to new-entrant provisions in the EU ETS, which can affect investment decisions by altering the economic ranking of possible alternative investments.[103] A second possible ("behavioural") explanation relates to operators (firms) failing to recognise the opportunity cost of using free allowances for compliance.

G) The EU ETS was generally internalised by firms (though not the driver, it was at least one variable in investment decisions) and it successfully induced organisational innovations. Moreover, credible long-term emission reduction targets are important triggers for low-carbon investment because they reduce regulatory uncertainty.

Some of the provisions in the recent reform for Phase IV (notably, the tightening of the cap and the establishment of the Innovation Fund) and the Market Stability Reserve address full on the weaknesses of the EU ETS that are most relevant to dynamic efficiency. These measures are consistent with the lessons that it has been possible to learn from the existing empirical evidence, though whether they will prove effective is, looking into the future, difficult to tell. Importantly, however, a well-established view in the literature of environmental innovation policy does not seem to find sufficient space in the EU's climate and energy strategy. This view, and recommendation, is that support to R&D should be strong and complementary to market-based instruments (Popp, 2010b). There is a compelling economic case, related to innovation spillovers, scale and network economies, competitiveness preservation and energy security, for complementing the EU ETS with stronger R&D policies. In the EU, spending on direct

[103] A recent contribution on this specific issue is by Flues and van Dender (2017).

support for renewable energy sources at the deployment level has so far dwarfed R&D support (Dechezleprêtre and Popp, 2015).

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Appendix

Table A1 – Summary of non-econometric studies on the electricity sector.

Study	**Data**	**MS**	**Sector (Year survey/interview)**	**Research Question**	**Main Conclusions**	**Reported ETS effect (1)**	**Kind of innovation identified (process/product)**	**ETS influence level**
Hoffmann (2007)	Primary. (interview to 5 companies (80% sectorial CO_2))	DE	Electricity (2006)	Which way does the ETS affect technology investment decisions that reduce CO_2 emissions?	- ETS drives small scale investments with short amortization times: retrofits (key to this was malus rule of the German's NAP), limited on portfolio choices (fuel choice of new plants), and limited effect on R&D investment: - Investment on renewable NOT driven by ETS - Increased interest on CCS related R&D, however considered too risky given carbon price level and volatility. Public perception is a more powerful driver. - ETS did NOT affect fuel choice for new plants in the portfolio (other factors more important) - Organizational changes to control the associated risk with investments decisions under the ETS - Effort: 2-10M per investment (amortization 2-5 years)	- CCS R&D - Retrofits	- Organizational (ETS risk management)	- Low to moderate
Rogge and Hoffmann (2010)	Primary (42 interviews)	DE	Electricity (2006-2009)	Which changes in the electricity's innovation system have been triggered by the ETS? (changes in innovation system include changes in: Knowledge and technologies, actors and networks, institutions and demand)	- Accelerates investment in new and existing plants (specially in coal-fired plants). - Significant impact on R&D in CCS technologies (also seen as a way to protect investment in existing plants). Strong effect for R&D linkages regarding CCS between large utilities and technology providers. - Additional driver for R&D in Energy efficiency (seen as the low risk option). - Feed in tariff systems are more relevant for R&D in renewables. - Incremental increase in the optimal efficiency level of new and existing plants	- R&D - retrofits	- Process innovation (increase Energy efficiency, R&D on CCS)	- Moderate impact

Rogge, Schneider, Hoffmann (2011)	Primary In depth interviews to 61 experts from 19 power generators, technology providers and project development)	DE	Electricity (2008-2009, when phase II was already determined and phase III details publicly available)	Which is the effect of the ETS on -R&D, -Adoption and -organizational change in the German power sector?	- The EU ETS has led to a significant intensification of RD&D activities through three routes that can be differentiated in whether the CO2 is captured before (IGCC), during (oxyfuel) or after combustion (post-combustion). - Strong increase in corporate CCS research. - ETS effects on R&D for Energy efficiency improvements: high for coal plants (accelerated previously ongoing R&D), very little for gas plants and wind turbines. - ETS effects on the adoption of new plants: For coal plants, there were perverse incentives before full auctioning. For gas plants no effect was driven by ETS. -ETS effects on retrofit of existing plants: For coal, the ETS contributes to increase retrofit activities.	- R&D - Retrofits (strong for coal plants)	-Process innovation (energy efficiency for coal plants) -Organizational innovation	- Overall low effect, but significant effect on R&D (CCS), on retrofit activities for coal plants and on organizational changes
Schmidt, Schneider, Hoffmann (2012)	Primary (201 firms: 65 power generators and 135 technology providers. Cluster analysis)	7 EU-MS	Electricity (2009)	"How do firms with diverse characteristics differ regarding their contributions to low-carbon technological change in the power sector?"	- Results obtained casts doubts upon whether the current policy mix is able to trigger the needed acceleration and redirection of Technological change in the magnitude needed to meet the targets: - Many firms appear to be behaviourally unaffected by climate regulations (BAU cluster with 40% of the firms is the bigger cluster identified).	-…	-…	-low impact
Hervé-Mignucci (2011)	Secondary data Survey of investments by top 5 carbon constrained utilities. Survey based on official corporate communications.	EU	Electricity (2004-2009)	How the ETS influenced longer-term impacts considering both operating Investments and financial investments in power generation?	- Since the ETS came into force, some investments made are clearly in favor of mitigating emissions (although this cannot be attributed to the ETS). - "The tighter constraint in phase II and expected phase III triggered clear investment related responses: highly carbon-emitting plants cancelled in favor of plants emitting less CO2 or regulated entities fully using offset project mechanisms to foster investments in lower carbon power plants."	- Cancellation of projected highly emitting plants.	- process innovation	- moderate effect (because of plant cancellations identified)

Notes: (1) Only reported innovation activities that according to the study were totally or partially induced by the ETS.

Table A2 – Summary of non-econometric studies on sectors other than electricity.

Study	**Data**	**MS**	**Sector (Year survey/interview)**	**Research Question**	**Main Conclusions**	**Reported ETS effect (1)**	**Kind of innovation identified (process/product)**	**ETS influence level**
Pontoglio (2008)	Primary (survey with 38 respondents)	IT	Paper Industry (2006)	Which is the role of EU ETS as driving factor for eco-innovations?	- On the EU ETS effect on innovation decisions: The majority of the respondents (52%) declared not to have realized/planned to introduce technological innovations aimed at reducing CO2 emissions. The remaining 48% already realized or developed projects for the coming years. As for the innovation typology, 38% were energy-efficient solutions; 24% were directed at the optimisation of the production process. - Italian paper plants are old on average, and hence there are significant margins for improving efficiency (in energy and the use of raw materials). -Energy prices are more relevant in driving energy efficiency (as carbon price is more uncertain)	-…	-…	- Low or null impact
Tomas et al. (2010)	Primary (Survey with 4 representative firms)	PT	Chemical Industry	How did the ETS affect the cost structure of the Portuguese chemical industry?	- The ETS has a limited impact: the costs increases observed were found to be non-relevant when compared to other EU industrial sectors. - Grandfathering and low prices cannot act as a stimulus for innovation	-…	-…	- low or null impact

Anderson, Convery, Di Maria (2011)	Primary (Mail interviews to 27 firms, 70% of the Irish allowance allocation)	IR	All regulated sectors	How did to the ETS affect technological changes in terms of: -machinery and equipment adoption, -process or behavioural change using existing equipment, -fuel switching and -R&D investments	- 48% of the respondents report technology adoption. However the ETS, despite considered, not the main driver (rising energy prices more important) - 74% report some form of process/behavioural change with current machinery, of which 30% admit some influence (marginal or strong) driven by ETS. - 41% engaged in fuel switching "in order to reduce carbon emissions" - 46% report that the ETS had influenced the way in which capital and investments are analysed.	- technology adoption - behavioural change - fuel switching	- Process innovation (some technological adoption, process changes and fuel switching)	- moderate impact
Rogge et al. (2011)	Primary (survey among paper producers (19) and technology providers (17))	DE	Paper Industry (2008-2009)	How did the ETS affect the innovation activities of: - R&D, - Adoption (investment in new plants or modernization of existing ones) and - Organizational change (procedural, structural or vision change) in the German Paper Industry?	- Innovation activities are mainly driven by market forces and hardly affected by the ETS. - ETS has low relevance for R&D investment: It is considered the least relevant factor (after the prices of paper, raw material and fuel, and after public acceptance). 65% of paper producers consider ETS has low relevance, 21% high relevance (lower for tech-providers). - Market factors more relevant for adoption. Although 37% considered ETS of high relevance in adoption decisions. - Regulated companies have realized more CO2-related organizational changes than non-regulated companies have.	- R&D (low or null) - Adoption (low or null) -Organizational (moderate)	-Process innovation (mainly organizational)	-Null or very low impact.
Gasbarro, Rizzi, Frey (2013)	Primary (6 case studies that include 24 installations)	IT	Pulp and paper Industry (2010-2011)	How did the ETS influence companies' Environmental Management Systems and to what extend did it affect investment planning of the pre-existing EMS progam?	- Companies having an EMS better integrate GHG emission monitoring and compliance within the existing organization. - But cost reduction (and factors other than the ETS) is the main driver of investment decisions. - "Investments in technological innovation to reduce carbon emissions are still limited, and investments tend to be mainly focused on market-available technologies for core processes." In any case, no driven by the ETS) - "the EU ETS has not been able to trigger additional investment in technological innovation, despite the presence of an EMS"	-…	- …	- Null effect

Gulbrandsen and Stenqvist (2013)	Primary (2 case studies)	SE, NO	Pulp and Paper Industry (2012)	How have the EU ETS influenced the climate strategies of two Nordic pulp and paper companies?	- ETS had a rather limited effect on the climate strategies. - EUA price is too low and allowances too abundant to underpin investment decisions. - However, increase of energy prices (perceived as the strongest influence of the ETS) have reinforced some long-term commitments to improve energy efficiency and reduce emissions, i.e. greater interest for own power assets.	-…	- …	- low or null effect
Borghesi et al (2015b)	Primary (29 Interviews to industry associations, and other experts or *industry* representatives)	CZ,PL, IT, DE, UK, ES, FR, NL	6 sectors: energy, ceramics, cement, paper, coke and refinery (2013)	Did policy support drive eco-innovations in terms of (i) technological and (ii) organizational innovations over the period 2000-2012?	- Energy: ETS did not contribute much to technology innovation in large combustion plants, but promoted fuel switch, CCS related innovations and organizational innovations - Ceramics and cement: innovations driven by market factors, not policy. - Paper: innovation driven by energy costs. Possible incremental influence of ETS on already ongoing innovation projects - Coke and refinery: Past policies were more relevant for innovation than current ETS.	- fuel switching - CCS projects	- process innovation (energy) - organizational innovation (energy)	- Moderate effect (energy), - very low or null effect (the remaining sectors)
Venmans (2016)	Primary (in-depth interview to 16 managers)	BE	Ceramics, lime and cement (2012)	How do managers perceive free allocation of allowances over emissions in terms of incentives to invest in abatement?	-The vast majority of them thought that allocation below emissions was a greater incentive to invest. "Companies combusting natural gas stated the availability of excess permits as the main reason for not including the carbon advantage of efficiency investments when assessing payback values." - Price uncertainty was perceived as a risk to be minimized and hence an incentive for low carbon investment." avoiding uncertainty from the EU ETS was an important motivation to invest in energy efficiency, certainly for future projects when companies are likely to be allocated below emissions". However, when the price volatility is faced directly, certain price vs uncertain price, most perceived uncertainty as a disincentive.	-…	-…	-…
Petsonk and Cozijnsen (2007)	Secondary data Case studies: description of	NE, DE FR	All sectors	How does the ETS stimulate the technological and process innovations?	- CO2 from a refinery is used by horticulture farmers, who otherwise would need to burn gas (to generate fertilizer). This already known technology only became economical once carbon had a price.	- …	- process innovation	- strong effect

	three cases that were possible because of the ETS existence				- Methane captures and generation of renewable energy are becoming more profitable thanks to the ETS.			
Fontini and Pavan (2014)	Secondary data (173 Italian paper plants)	IT	Paper and pulp industry (2005-2010)	Decomposition of the variation in emission in terms of composition, technic and scale effects.	- During phase I there was a reduction emission via composition effect: shift in production towards products that cause less carbon emissions. This trend could be driven by either change in production or change in demand. - During phase II technological change played a contribution, but the scale effect (decrease in overall output) always dominate emission reduction, in both phases I and II.	-…	-…	- low effect (potentially only, as the method does not identify the ETS effect)

Notes: (1) Only reported innovation activities that according to the study were fully or partially induced by the EU ETS

Chapter 5

The international dimension

1. Introduction

While remaining the largest Emissions Trading System (ETS) in the world (till China's national ETS comes into operation[104]), the EU ETS covers about 5% of global greenhouse gas (GHG) emissions. This means that the significance of its impact on global emissions critically depends on whether its implementation can bring about climate action outside Europe too. From the very beginning, the EU ETS has not only been the prime instrument for decarbonising the European economy, but it has also played a wider role at the international level. Throughout its existence, it has been a building block of the international carbon market as well as a reference for other jurisdictions with similar policies in place or under consideration. This wider role of the EU ETS serves well the EU's ambition to lead the global fight against climate change. What is more, whether such ambition is fulfilled also depends on how the EU ETS performs with respect to its own objectives. As the EU ETS is still a relatively novel policy, its being effective is central for its own credibility and for that of the EU as a leader in the fight against climate change.

Despite its relevance, the international dimension of the EU ETS and, in general, of any ETS is admittedly a rather blurred concept[105]. Indeed, to the best of our knowledge, the international dimension of an ETS as such has not been conceptualised in the literature. Thus, to delimit the scope of our analysis, and for general clarity, we have coined the following definition:

The international dimension of an ETS pertains to the capacity of its regulation to produce economic or environmental effects outside its jurisdiction and, the other way round, to its being subject to similar effects of climate policies abroad, whether through a linkage with those or without.

[104] Launched in December 2017, China's national ETS is expected to be operative by 2020. The emissions coverage of the Chinese ETS will be around twice that of the EU ETS (ICAP, 2018).
[105] In this chapter, as in most contexts, ETS is used as a synonym of cap-and-trade system.

This definition encompasses two ways in which the regulation of an ETS may have environmental or economic effects beyond its jurisdiction. One operates through a linkage with other ETSs or other climate policies overseas. Following Mehling *et al.*'s (2018a) taxonomy, the linkage between two or more climate policies can be "soft" or "hard" depending on whether it connects the respective (explicit or implicit) carbon prices or, in addition, it allows for emission reductions realised in one jurisdiction to be used for compliance in another. The other way in which an ETS may produce effects outside its jurisdiction is by merely influencing – without a linkage – existing climate policies abroad or the likelihood of climate action being undertaken abroad. For example, an ETS could become a model to follow, so that its implementation may inspire regulatory changes in already existing ETSs or encourage the adoption of new ETSs or other climate policies (Wettestad and Gulbrandsen, 2018).

The reason why linking climate policies is generally a possibility worth considering is that, by establishing a more homogenous carbon price across sectors or jurisdictions, it reduces the total cost of cutting emissions by a certain amount. That is, in principle, it produces a gain in terms of aggregate economic efficiency. For any linkage to happen, however, the sharing of the efficiency gain, between and within the given jurisdictions, needs to be perceived as fair by all parties. Deciding on whether to link an ETS to other climate policies indeed can be a complicated question. Policymakers will examine it taking into consideration different factors and in the light of multiple policy objectives (Flachsland *et al.*, 2009). Further, if any form of linkage is pursued, deep preliminary cooperation between the authorities involved is necessary. Things do seem to be moving in this direction. International cooperation aimed at fostering or aligning domestic climate policies is, arguably, higher than ever on the global policy agenda.

The need to circumscribe the topic by first devising a definition of "international dimension of an ETS", in abstract terms, is not the only peculiarity of this chapter compared to the previous ones. The present chapter is also characterised by some greater space devoted to relevant historical developments and to relevant theoretical studies. This is not so much a deliberate choice as a necessary one. First and foremost, any analysis of the international dimension of the EU ETS cannot ignore the nature or the state of the international climate change regime. This context has radically changed over the past two decades in passing from the Kyoto Protocol's regime to the Paris Agreement's, with a few years in between characterised by high uncertainty about the future setting. Opportunities and to some extent even motives for integrating, aligning, or just for promoting carbon markets and other climate policies, have evolved in connection with those developments. Second, though the EU ETS has exerted much influence on the outside world, only few experiences have provided suitable material for empirical analyses. As a result, the relevant empirical literature is rather small and limited in terms of topics covered. The theoretical literature is, by contrast, large and diverse. It spans questions concerning the incentives for international climate cooperation and the related political economy (clubs literature), the different forms of carbon market integration and the respective economic implications (linking literature) and, more recently, the processes of carbon market cooperation and carbon market diffusion.

The chapter is organised as follows. Section 2 recalls the evolution of the international climate change regime. Section 3 reviews the role of the EU ETS in the international carbon market so far. Section 4 illustrates the club approach to climate cooperation. Section 5 concludes.

2. The international climate change regime: from Kyoto to Paris[106]

Knowing the context in which an ETS operates, most notably the international climate change regime, is essential for understanding its international dimension. The EU ETS was established in connection with the Kyoto Protocol. Then, after a phase of uncertainty about the future international climate change regime, it recently entered the Paris Agreement's era.

2.1 The Kyoto Protocol's regime and the gestation of its successor

Adopted in December 1997 and in force since February 2005, the Kyoto Protocol to the United Nations Framework Convention on Climate Change (UNFCCC) has been the first international agreement imposing binding targets for GHG emissions. Under the Kyoto Protocol, 37 industrialised countries and the EU committed to reducing their emissions by an average of 5% against 1990 levels, over the period 2008-2012. Each country had its own emissions target (EU: -8%; Japan: -6%: Australia: +8%; etc.[107]), but – crucially – these targets had been defined through a negotiation process rather than determined by each country unilaterally. The characterisation of the Kyoto Protocol as a top-down regime mainly relates to this procedure, whereby the distribution of efforts – not only the overall climate mitigation objective – among the Parties (countries) is agreed from the outset.

[106] This section outlines the evolution of the international climate regime (from reflecting a top-down approach to a "hybrid" approach) and, within it, the role of market-based mechanisms. For a comprehensive account of the history of the international climate regime, till 2012, see Gupta (2014). For analyses of the Kyoto Protocol and of the Paris Agreement, see Grubb (2003) and Falkner (2016), respectively. To understand how we got to the Paris Agreement, see Bodansky (2011) and Victor (2011).
[107] The national emissions reduction targets are found in Annex-B of the Kyoto Protocol.

The Kyoto Protocol allowed countries a certain degree of flexibility in meeting their climate mitigation commitments through three market-based mechanisms: International Emissions Trading (IET), the Clean Development Mechanism (CDM) and Joint Implementation (JI). Cost-effectiveness is the economic principle underpinning the three mechanisms, which share the rationale "cut emissions where is cheapest". For Parties with emissions targets, the so-called Annex-B countries, IET allows exchanges of Assigned Amount Units (AAUs) made available by those overshooting their targets[108]. The CDM and JI, by contrast, generate emission offset credits on a project basis, that is, against individual projects which are proved to have reduced emissions. The CDM involves investment in sustainable development projects that reduce emissions in developing countries, whereas JI enables industrialised countries to carry out joint implementation projects with other developed countries (most often countries with economies in transition). Together with AAUs and Removal Units (RMUs) generated by land use, land-use change and forestry (LULUCF) activities, the credits generated by the CDM and JI, called Certified Emission Reductions (CERs) and Emission Reduction Units (ERUs), respectively, are the tradable Kyoto units.

While based on sound economic principles, the Kyoto Protocol had the major limitation to constrain less than 30% of global GHG emissions. This is why, ever since the Kyoto Protocol's entry into force, the central question facing the UN climate change regime was what to do after 2012, when initial limits on GHG emissions would expire (Bodansky, 2011). The EU was strongly in favour of a Kyoto-type regime, this time inclusive of emissions targets also for developing countries. Support for this solution from other countries, however, clearly was not sufficient (Bäckstrand and Elgström, 2013). The 15th UNFCCC Conference of the Parties (COP15), held in Copenhagen (Denmark) in December 2009, represents at the same time a low and a turning point in international climate change negotiations. The expected outcome of the conference was originally a new legal agreement addressing the post-2012 period. Such an agreement, however, did not materialise. Instead of it, COP15 produced a brief wide-ranging political agreement. Most importantly, the Copenhagen Accord started a bottom-up process that allowed each country to define its own commitments and actions unilaterally (Bodansky, 2011).

After COP15, international negotiations moved steadily toward the creation of a climate change regime informed by the bottom-up approach. Building on the Copenhagen Accord,

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[108] Emissions targets are expressed as levels of allowed emissions, or assigned amounts, over the commitment period. The allowed emissions are divided into AAUs.

COP16, in Cancun (Mexico), was decisive in accelerating things in that direction. At COP17, in Durban (South Africa), among other significant results was the decision to adopt by 2015 (COP21) a new agreement for the post-2020 period. The future agreement would include unilaterally-determined emissions targets for all countries. In the meantime, the Kyoto Protocol's second commitment period (2013-2020) would act as a bridge to the post-2020 regime. However, countries that agreed to new emissions commitments for the period 2013-2020 represent less than 15% of global GHG emissions[109].

2.2 The Paris Agreement

Adopted in December 2015 and in force since November 2016, the Paris Agreement is the legal foundation of the international climate change regime post-2020. The Paris Agreement was signed by 195 countries, accounting for 99% of global GHG emissions, and subsequently ratified by almost all of those, with the resounding defection of the United States[110]. The ultimate objective of the Paris Agreement is to keep the increase in the global average temperature to "well below" 2°C above pre-industrial levels ("pursuing efforts to limit the temperature increase to 1.5 °C above pre-industrial levels"). Effectively, it sets out a long-term framework for addressing the climate change problem in a definitive way.

The Paris Agreement radically differs from the Kyoto Protocol in several respects. First and foremost, the Paris Agreement obliges all countries to take action to limit global warming, thereby overcoming the main limitation of the Kyoto Protocol. Second, each country's contribution to the collective effort of global climate stabilisation is determined unilaterally. This aspect is emphasised by the locution Nationally Determined Contributions (NDCs), which is how countries' voluntary commitments, or formal pledges, are called. Therefore, Parties did not have to preemptively agree on a burden sharing, which proved to be the main problem in previous attempts to establish a new Kyoto-type regime. Third, the NDCs can be heterogeneous, which means that they do not necessarily have to be specified in terms of emissions reduction targets[111]. Fourth, the Paris Agreement is inherently dynamic, in that the

[109] While Canada withdrew from the Kyoto Protocol already in 2011, other major emitters like Japan and Russia decided not to take up new emissions reduction commitments for the 2013-2020 period. The principle of a second commitment period of the Kyoto Protocol was agreed at COP17 in Durban, in 2011, and formalised at COP18 in Doha (Qatar), the year after. At the time of writing, however, the Doha Amendment to the Kyoto Protocol has not yet come into force because the number of countries having ratified the Amendment (122) falls short of the minimum required (144).
[110] At the time of writing (December 2018) the Paris Agreement has been ratified by 184 countries.
[111] For example, the first NDC committed by the EU is "a binding target of an at least 40% domestic reduction in greenhouse gas emissions by 2030 compared to 1990". The first NDC committed by China is articulated as follows: "To achieve the peaking of carbon dioxide emissions around 2030 and making best efforts to peak early;

achievement of its climate mitigation objective rests on the unfolding of a "ratchet mechanism" whereby the NDCs are reviewed and increased at five-year intervals. Indeed, the initial NDCs taken together fall well short of the target (UNEP, 2017). The Paris Agreement as a whole has been described as a hybrid approach blending bottom-up and top-down features (Chan *et al.*, 2018). The first relate to the freedom of Parties in setting their NDCs. The second refer to the obligations that Parties have, such as monitoring and reporting duties among others[112].

The Paris Agreement is the successful outcome of a process originated from the diplomatic failure of the UNFCCC COP15 (2009), in Copenhagen[113]. At the same time, it is the beginning of a radically different regime that tolerates some important unknowns about its own effectiveness in addressing climate change. Both perspectives are true, which probably partly explains the existing range of views over the Paris Agreement: regarded by some as "Plan A" (Sabel and Victor, 2016) and, by others, as being entirely inadequate for the reality of climate change (Spash, 2016)[114]. As Keohane and Oppenheimer (2016) put it, "the Paris Agreement is less an accomplishment than part of an ongoing process".

2.3 Cooperative Approaches in the Paris Agreement

A central feature of the Paris Agreement, particularly relevant to this chapter, is the possibility for Parties to use so-called Cooperative Approaches in order to meet their NDCs. Though they will translate into mechanisms presenting new features, the Paris Agreement's Cooperative Approaches are the direct descendants of the Kyoto Protocol's Flexible Mechanisms[115]. Article 6 of the Paris Agreement describes them with a rather vague wording, leaving much space for interpretation over how exactly they will work. At the time of writing, basic issues such as scope, governance and infrastructure of the Cooperative Approaches are being negotiated

To lower carbon dioxide emissions per unit of GDP by 60% to 65% from the 2005 level; To increase the share of non-fossil fuels in primary energy consumption to around 20%; and To increase the forest stock volume by around 4.5 billion cubic meters on the 2005 level." All submitted NDCs can be found on the UNFCCC website.

[112] For a legal analysis of the Paris Agreement, see Bodansky (2016).

[113] The idea of a bottom-up polycentric approach to address climate changes precedes COP15. See, e.g., Ostrom (2014).

[114] Sabel and Victor (2016) hail the Paris Agreement not as "a consoling alternative to failure […], but as a superior way to coordinate action in the face of massive uncertainty about the interests, capabilities, and intents of the key players. It is not a time-consuming detour on the way to the main goal but rather the only viable path to achieving radical transformations in national policies that lead to deep cuts in emissions".

[115] For a detailed comparison of market-based mechanisms in the Paris Agreement and in the Kyoto Protocol, see Schneider *et al.* (2016).

(Asian Development Bank, 2018)[116]. In the meantime, many analysts have expressed their own views[117].

Article 6, whose text is reported in the Appendix, comprises four main parts:

1) Article 6.1 contains a general provision for international cooperation towards Parties' NDCs. It is broad and meant to cover all specific cases of cooperation in Article 6, and others that may emerge in the future.
2) Articles 6.2 and 6.3 contain provisions for Parties to cooperate through internationally transferred mitigation outcomes (ITMOs). These articles specify what Parties shall do to use ITMOs towards their NDCs. The formal intervention by the Conference of the Parties serving as the meeting of the Parties to the Paris Agreement (CMA) is limited to developing and providing guidance. The only mandated role for the CMA under 6.2 is to develop accounting standards that any Parties engaged in ITMOs would have to observe. In this respect, Parties are given a significant amount of freedom in what they can do[118].
3) Articles 6.4-6.7 contain provisions which outline a multi-scope mechanism to produce mitigation outcomes that can be used to fulfil the NDC of another Party and support sustainable development. In contrast to 6.2, this mechanism is designed under the authority and guidance of the CMA. The CMA has the role of setting standards in all aspects, including approval processes, technical aspects for quality and quantity of what is being transferred, and avoidance of double counting[119].
4) Articles 6.8 and 6.9 establish a framework of non-market approaches. It is still largely unclear what will be covered under this part of Article 6, but some focus, e.g. the coordination of different non-market cooperative approaches, is starting to emerge (Asian Development Bank, 2018).

The two mechanisms proposed in Articles 6.2-6.3 and Articles 6.4-6.7 are meant to allow for international emissions trading and to create a new and improved Kyoto-type CDM (Marcu, 2016a). Article 6.2 provides a framework where innovative ideas can be developed, whereas

[116] A draft rulebook for the operationalization of the Paris Agreement has been adopted at the UNFCCC COP24, held in Katowice (Poland), in December 2018. The draft rulebook includes guidance on the Cooperative Approaches referred to in Article 6 (see: https://unfccc.int/documents/187593).
[117] Among others: Bultheel *et al.* (2016); Carbon Market Watch (2016); Dahan *et al.* (2016); Carbon Market Institute (2017); Olsen *et al.* (2018); Marcu (2016a, 2016b, 2016c, 2017); Asian Development Bank (2018).
[118] For a deep reflection on Article 6.2, the reader is referred to Marcu *et al*. (2017) and Marcu and Zaman (2018).
[119] The mechanism is commonly referred to as Sustainable Development Mechanism (SDM). Marcu (2016a, 2016b) suggests an alternative name: Sustainable Mitigation Mechanism would be more appropriate, as it would emphasise the dual purpose of contributing to mitigation and supporting sustainability.

Article 6.4 provides a solid, but possibly more restrictive approach (Marcu, 2016c). Crucially, Articles 6.2 and 6.3 open the door to linking ETSs, or, indeed, any market-based mechanisms[120].

Parties that have an interest in exploring Article 6 may differ in terms of the carbon market development at a national level. Different options exist for them to make use of the mechanisms implied in Articles 6.2 and 6.4. For example, a country that has already developed domestic market mechanisms, such as an ETS or a sectoral market-based scheme, may be interested in using ITMOs from Article 6.2 to link its ETS with another ETS at the international level. A country that has gained experience with the CDM and JI, may explore possible options under Article 6.4 (Cames *et al.* 2016; Olsen *et al.*, 2018).

Importantly, the Paris Agreement rests on the use of the Cooperative Approaches as a means to facilitate the ratchet mechanism. The logic is: cooperation allows reducing the cost of meeting the NDCs, thus creating space for increasing ambition in mitigation. As the sum of the initial NDCs falls (well) short of the Paris Agreement's mitigation objective, cooperation as an enabler of the ratchet mechanism is a defining characteristic of the Paris Agreement. By reducing the cost of meeting the NDCs, cooperation has the potential to induce greater mitigation efforts[121].

3. The EU ETS in the international carbon market

Both the configuration and the size of the international carbon market are closely related to the configuration and the state of the international climate change regime. With its Flexible Mechanisms (IET, CDM and JI), the Kyoto Protocol enabled the emergence of a large international carbon market. The fate of the same market, however, followed the anticlimax of the Kyoto Protocol's regime. The outcome of the UNFCCC COP15, in 2009, certified the end of all hopes for a new extended Kyoto-type regime post-2012, that is, one with legally binding emissions targets for all (or almost all) countries. Consequently, the international carbon market was first plagued by uncertainty over the future climate change regime. It then landed in the desolate territory of the Kyoto Protocol's second commitment period (2013-2020). The Paris Agreement, thanks to its Cooperative Approaches (Article 6) and almost universal participation of the international community, has revived the prospects for the future of the international carbon market. The EU is in the position to actively contribute to this likely

[120] Mehling *et al.* (2018b) illustrate how also climate policies other than ETSs could be linked under Article 6.
[121] It has been estimated that, by the middle of this century, an international carbon market has the potential to reduce global mitigation costs by over 50% (WB, Ecofys and Vivid Economics, 2016).

renaissance, capitalising on some important experiences accumulated through the EU ETS in the past decade or so.

3.1 Experience with the Kyoto credits

Since its inception, the EU ETS was designed so as to be part of the nascent international carbon market and thereby to contribute to its development. In concrete terms, the EU ETS was directly connected to the Kyoto system through the so-called Linking Directive (European Parliament and Council, 2004)[122], which allowed the owners of regulated installations to use international emissions offset credits generated by the CDM and JI to meet part of their compliance obligations[123]. The main sources of demand for tradable emission units under the Kyoto Protocol, then, were *a)* the governments of Annex-B countries and *b)* the owners of installations in the EU ETS. The first were issued AAUs and, at the end of the Kyoto Protocol's first commitment period (2008-2012), had to surrender AAUs or CERs/ERUs equal to their economy-wide emissions. The second were issued European Union Allowances (EUAs) and, every year, had to surrender EUAs or (subject to limitations) CERs/ERUs matching their annual emissions[124].

The importance of the EU ETS in terms of market volume, strength of demand for Kyoto trading units (as compared to that of Annex-B governments) and, therefore, level of carbon prices, meant that the EUA price soon became the reference price of the international carbon market. In theory, without restrictions on the use of international credits in the EU ETS, CER/ERU prices would have converged to EUA prices. In reality, however, CER/ERU prices tended to be a few euros lower than EUA prices. The discount reflected the risk that CERs/ERUs in the secondary market may not be usable for compliance in the EU ETS. This risk, in turn, was mainly related to uncertainty as to whether the limit for the use of international credits in the EU ETS would be reached.

Indeed, as of Phase II (2008-2012), quantitative restrictions to the use of international credits were applied. Specifically, installation owners were allowed to use international credits up to a percentage of total allowances determined in the National Allocation Plans. These percentages ranged from 0% (Estonia) to 20.6% (Spain), summing to 13.4% for all Member States taken together (Ellerman *et al.*, 2010). The possibility to use international credits was

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[122] The Linking Directive was passed as an amendment to the ETS Directive.

[123] European Union Allowances issued under the EU ETS were matched one to one with AAUs from the issuing Member State.

[124] For a detailed explanation of the relationship between the EU ETS and the international carbon market, see Ellerman *et al.* (2010).

further tightened in Phase III, through both quantitative and qualitative restrictions[125]. To be sure, low carbon prices in the EU ETS did not invite the European legislator to open up the market any further to inflows of international emissions offset credits cheaper than EUAs, as this would have caused additional downward pressure on EUA prices. At the same time, the use of CERs originated from projects involving HFC-23 gas destruction and nitrous oxide (N_2O) emissions from adipic acid production was banned due to serious environmental integrity concerns (European Commission, 2011).

All in all, the linkage of the EU ETS to the Kyoto Flexible Mechanisms can be regarded has having produced some positive results but also important criticalities for the European carbon market. On the one hand, the linkage has delivered the intended result of extending the carbon price signal to countries and sectors not covered by the EU ETS, leading to the development of emissions reduction projects that otherwise would have not taken place[126]. This is particularly relevant for projects undertaken in developing countries, as they imply technology transfers as well as institutional familiarisation with carbon markets. On the other hand, the inflow of international offset credits affected the EU ETS in two main ways. One relates to the dubious nature of some emission reduction projects for which CERs were issued. The two said cases of projects involving HFC-23 gas destruction and adipic acid production justified the intervention of the European Commission. In general, for any ETS, the importance of preserving its environmental integrity could not be overemphasised.[127] The second criticality concerns the impact that the inflow of emission offsets had on carbon prices, putting further downward pressure on already low prices. In addition, the quantitative limits on emission offsets resulted in price differentials with the allowances issued under the EU ETS, enabling substantial windfall profits (de Bruyn *et al.*, 2016).

3.2 Linking with other ETSs

In addition to the direct link with the Kyoto Mechanisms, other experiences have been relevant to the international dimension of the EU ETS so far. Notably: the incorporation of Norway, Iceland and Liechtenstein into the EU ETS; processes of linking with other ETSs; and the

[125] Installations which already fell into the scope of the EU ETS in the period 2008 to 2012 may use credits in the period 2008-2020 up to a limit of 11% of their allocation for 2008-2012. For new entrants, the limit is 4.5% of verified emissions over 2013-2020 (European Commission, 2015).
[126] The use of CERs/ERUs for compliance in the EU ETS does not result in greater overall emission reductions. The offset system only affects the location of emission reductions.
[127] In relation to ensuring the environmental integrity of an ETS, assessing the additionality of emission offsets is one of the main technical challenges and one that will continue to be much debated (Schneider and La Hoz Theuer, 2018; Cames *et al.*, 2016; Arup and Zhang, 2015).

participation of the European Commission in international cooperation programmes supporting the adoption and diffusion of carbon pricing.

Since its establishment, the EU ETS has expanded in the coverage of countries (our focus here), sectors and GHGs. In 2007, Bulgaria and Romania were incorporated into the EU ETS merely as a consequence of their becoming EU members. The following year, after extended negotiations, Norway, Iceland and Liechtenstein also joined the EU ETS. In this case, the three countries, which were not EU members, agreed to join the EU ETS by virtue of being partners of the EU in the European Economic Area. Norway had already its own national ETS, which had been in operation since 2005. Since the European Commission rejected Norway's plan to link with the EU ETS, Norway's ETS was fully integrated into the EU ETS. Full integration required limited regulatory adjustments in the Norwegian ETS (its design was similar to that of the EU ETS) and, on the EU side, only modest temporary concessions in the application of the EU ETS legislation (Ellerman *et al.*, 2010).

In 2010, negotiations started between the EU and Switzerland to link the respective ETSs. The agreement on a two-way (or bilateral) linking was signed in 2017 and is currently awaiting the final vote of the two parliaments for its ratification. The link will allow participants in the EU ETS to use allowances from the Swiss ETS for compliance, and vice versa. It is the first agreement of this kind for the EU and between two Parties to the Paris Agreement on climate change. In order to align their designing elements, the Swiss ETS will include the GHG emission from the aviation sector which at present is not capped by any specific legislative measure in Switzerland. Once the agreement is operational, prices should converge resulting in reduced competitive distortions, as companies have level playing field and simplified business especially for multinationals (Betz, 2017). Other expected benefits for the Swiss ETS include lower cost of emission reductions, enhanced liquidity and price stability (Vöhringer, 2012; Fuessler *et al.*, 2015). In return, the economic impacts on the EU ETS should be negligible given the considerable difference in scale between the EU ETS and the Swiss ETS (Englert, 2015)[128].

In 2012, the European Commission and the Australian government announced their intention to link their ETSs[129]. The first stage of linking was supposed to occur in July 2015

[128] The EU ETS regulates approximately 11,000 stationary installations which together emit around 2 billion tonnes of CO_2. The EU ETS also includes aircraft operators, which account for emissions of around 200 million tonnes of CO_2. Approximately 50 companies, which emit a total of around 5.5 million tonnes of CO_2, participate in the Swiss ETS (Swiss Federal Office for the Environment, 2015).

[129] Before the elections, the ruling government coalition in Australia adopted the Carbon Pricing Programme (CPP), which involved a carbon market starting in 2015. However, the party that won the 2013 elections cancelled the CPP.

with a one-way (unilateral) link, whereby covered entities in Australia would have been able to use EUAs to fulfil up to 50% of their compliance obligations. The linkage was expected to become bilateral three years later, when the EU regulated entities would have been allowed to use Australian Emission Units (AEUs) to fulfil their liabilities. The proposed link, however, raised concerns about the current over-allocation in the EU ETS. While the one-way link would have lowered compliance costs in Australia and helped to reduce the allowance surplus in the EU ETS, it might have appeared to weaken the ambition of the Australian system, creating financial flows from Australia to the EU. In the end, following Australia's national elections, in 2013, the linking plan was abandoned, for the new Australian government decided to repeal its ETS legislation.

3.3 International cooperation

The development of the international carbon market is a complex, lengthy process. Experience shows that integration of ETSs already in operation can take several years. It also shows that integration of the same ETSs can be vulnerable to political swings within the jurisdictions involved. To consider integrating ETSs, however, well-functioning domestic ETSs are needed in the first place. Establishing a well-functioning ETS indeed requires a nontrivial institutional effort.

Against this backdrop, the European Commission has been supporting other countries in the development of their ETSs by sharing its experience with the EU ETS. Notably, the European Commission is a founding member of the International Carbon Action Partnership (ICAP), which brings together countries and regions with mandatory ETSs or intending to adopt mandatory ETSs. The ICAP provides a forum for sharing experience and knowledge as well as training courses. The European Commission is also a contributing participant of the World Bank's Partnership of Market Readiness, which provides grant financing and technical assistance for capacity building and piloting of market-based tools for GHG emissions reduction. Furthermore, in 2017, the European Commission (in collaboration with the Florence School of Regulation) initiated the Florence Process. The Florence Process is a policy dialogue that brings together representatives of the key jurisdictions with carbon market policies, including China, Canada, California and New Zealand, to address issues of common interest (Borghesi *et al.*, 2018).

International cooperation is carried out on a bilateral basis too. The project "Platform for Policy Dialogue and Cooperation between EU and China on Emissions Trading" is currently being implemented. It aims to provide capacity building and training to support Chinese authorities in their efforts to implement and further develop the Chinese national ETS. At the EU-China summit in 2018, the EU and China signed a Memorandum of Understanding to further enhance cooperation on emissions trading. It provides a reinforced basis for the policy dialogue, including through further forms of cooperation such as the joint organisation of seminars and workshops, joint research activities and ad-hoc working groups. Similarly, the European Commission supports Korea through a technical assistance project focused on building the necessary capacity to implement Korea's ETS. The success of Korea's ETS, which was established in 2015, is particularly important as it could trigger the expansion of emissions trading among emerging economies and developing countries.

4. The club approach to climate cooperation

Climate change mitigation (or, equally, a stable climate) is one perfect example of global public good, in that the resulting avoided damages of climate change are a non-rivalrous and non-excludable good throughout the world[130]. The fundamental problem with public goods is the connected incentive that the agents involved face to free-ride on the efforts of the others; that is, not to contribute to the joint effort, or to contribute less than the social optimum requires (Ostrom, 2015). The historical inability of the international community to address climate change in an effective manner, as proved by the past two decades of UNFCCC negotiations, is rooted in this collective action problem. In direct response to it, the club approach to climate cooperation offers the most popular conceptual framework for fostering international cooperation on climate action.

4.1 Climate clubs

Club theory is a strand, or rather a niche, of economics whose origin is usually identified with a seminal study by Buchanan (1965). In general terms, a club can be characterised as a voluntary group whose members share a set of benefits, referred to as the "club good", from which non-members are excluded. In the climate policy context, climate clubs are cooperative models of climate action for overcoming the limits imposed by free-riding behaviour on the

[130] A good is non-rivalrous if its consumption by one consumer does not prevent its consumption by others. A good is non-excludable if non-paying consumers cannot be prevented from accessing it.

part of one or more agents. Climate clubs are understood to be started by small groups of enthusiastic agents (e.g., countries, individuals, businesses)[131], who would then try to entice other reluctant peers to join in (Victor, 2011). The expediency of climate clubs is closely related to the reliance of the UNFCCC on the consensus rule. In many situations this rule dampens the ambition of climate action, as it suffices only one or few Parties to block collective decisions entailing appropriate efforts (i.e., of the scale required by climate science). The club strategy involves limited-membership regimes that produce or secure economic or other non-climate benefits accruing to participants in return for their stronger mitigation action or other climate-change-related action (e.g., adaptation, climate engineering or climate compensation). Examples of such benefits are, among others, access to R&D or financial programs (Carraro, 2016) and access to preferential trade arrangements (Nordhaus, 2015). Furthermore, both conditional commitments (pledges) from the club members and financial transfers within the club can be effective "add-ons" – additional to the club good(s) – for creating and subsequently enlarging climate clubs (Sprinz *et al.*, 2018).

Though climate clubs in a strict sense, representing an alternative or a key complement to the UNFCCC, do not exist as yet, a broad understanding of climate clubs allows classifying different observed forms of climate cooperation under the same framework. In this regard, Stewart *et al.* (2017) distinguish between "classic clubs", "pseudo clubs", and "coalitions". The classification is based on the relevance of the club benefits and the degree to which non-club members can be excluded from their fruition. While classic clubs provide clear and readily excludable benefits, pseudo clubs provide benefits that are more diffuse, less readily excludable, and potentially less easily quantifiable. An example are the reputational benefits of companies participating in carbon measurement and disclosure programmes[132]. Importantly, however, pseudo clubs without government regulation will not move from monitoring emissions to enforcing emission reductions (Green, 2017). Climate coalitions are effectively further-diluted arrangements. They generally offer information- or publicity-related benefits, while requiring limited or no real environmental commitments (Weischer *et al.*, 2012; Andresen, 2014).

4.2 Carbon market clubs

[131] More precisely, "enthusiastic" here means willing to take action beyond what maximises one's material self-interest (Hovi *et al.*, 2016).
[132] E.g., the Carbon Disclosure Project (www.cdp.net).

Using the taxonomy above, carbon market clubs are a type of classic climate club. Their specificity is in emission trading being the mandated approach for cooperatively pursuing the mitigation objective. Writing at a time when the idea of a bottom-up international climate change regime had only started to emerge, Ellerman (2010) suggested that the development of international carbon markets necessitates the provision of club benefits beyond those connected to market participation. On the same question (commonly referred to in the literature as the question of "issue linkage"), Keohane *et al.* (2017) present a more optimistic view, which is perhaps more consonant with the new context of the Paris Agreement and the related expectations. The authors envision the formation of a club of linked carbon markets that countries would want to join without additional incentives. The club, as an autonomous institution, would need to: *a)* create the conditions for mutual recognition of emission units among members; *b)* maintain the market infrastructure necessary for trading; *c)* establish clear criteria for membership; and *d)* inform assessments of mitigation effort and ambition among current and prospective members. An historical example of how such a club could be initiated and operated is, from the realm of international trade, the General Agreement on Tariffs and Trade.

Though the formation of international carbon market clubs was perfectly possible already under the Kyoto Protocol, there is a general expectation that they will thrive in the Paris Agreement era. The simple fact that most countries in the world have committed to reducing their emissions, and many of them have declared to intend using carbon pricing (Marcu and Sugathan, 2018), implies that many more opportunities for linking ETSs should arise in the future[133]. Also, international linking of climate policies, including ETSs, will offer lower abatement costs. It will therefore facilitate the ratchet mechanism on which the Paris Agreement rests to achieve its ultimate objective of climate change mitigation. Furthermore, another way (at least) in which the Paris Agreement is expected to expand the opportunities for linking carbon markets is by accounting for emission reductions undertaken by sub- and nonstate actors (Bernstein and Hoffmann, 2018; Hale, 2016).

No matter how favourable the context related to the international climate change regime may be, carbon market clubs are not expected to materialise quickly, for integrating ETSs is generally a lengthy process. Deep cooperation between the relevant authorities is necessary to prepare and implement the linkage of the given ETSs and then to manage the new system. In

[133] Over 80 national governments have declared that they intend to use carbon pricing for meeting their NDCs (Marcu and Sugathan, 2018). Consistent with this revelation of policy preferences, the count of operating ETSs around the world has increased, reaching 21 (at different levels of government) in early 2018 (ICAP, 2018).

their "Guide to linking Emissions Trading Systems", Santikarn *et al.* (2018) identify three broad phases of the linking process: genesis, negotiation and implementation. In the first phase, the possibility of linking and the elements of a successful link are assessed. In the negotiation phase, authorities gain deeper understanding of the other ETSs, agree a structure for the negotiations and, then, define the details of the linking agreement. The implementation phase sees the operationalisation and launch of the linked market. Important elements of the process are *a)* the regulatory changes that may be needed to align the ETSs, *b)* the involvement of stakeholders, and *c)* the possible establishment of a new institution overseeing the linked market.

5. Conclusions

The international climate change regime has undergone major changes over the past twenty years. In many ways, the settings laid out by the Kyoto Protocol and by the Paris Agreement are two different worlds. The transition from one "world" to the other took several years, generating a prolonged phase of uncertainty. These changes have had implications for the international dimension of the EU ETS which, as a building block of the international carbon market, has gone through them all.

Since its inception, the EU ETS has exerted significant influence on the outside world. Under the Kyoto Protocol's regime, the recognition for compliance purposes of international emission offset credits generated by the CDM and the JI was the most important initiative. The results of this experience have been mixed. On the one hand, the linkage to the Kyoto Flexible Mechanisms has produced the intended, important outcome of extending the carbon price signal to countries and sectors not covered by the EU ETS. On the other, the inflow of international credits negatively affected the EU ETS, both through the dubious nature of the projects underlying some credits and by putting further downward pressure on already low carbon prices. The incorporation of Norway, Iceland and Liechtenstein is the second relevant experience in the early years of the EU ETS.

After COP15 (Copenhagen), despite the difficulties of the international negotiations, the EU did not give up its ambition to be a leader in the fight against climate change and to strengthen the international carbon market through the EU ETS. It first conceived a project for an OECD-wide carbon market, which was not pursued after the proposal for a national ETS was rejected by the US Senate. Then, bilateral negotiations were engaged with Australia and Switzerland to link the EU ETS with the ETSs of those countries. The first linkage eventually did not happen because the new Australian government was opposed to it. The one with Switzerland is currently awaiting ratification by the two parliaments. In the meantime, the EU has contributed to international carbon market cooperation by providing capacity building programmes for setting up and managing domestic ETSs.

The Paris Agreement has established a new international climate change regime that clearly promotes international climate cooperation. In many cases, this will take the form of carbon market cooperation. A result that the Paris Agreement has already produced is indeed the revival of positive prospects for the international carbon market. Expectations are that carbon pricing policies will spread further across the world and that carbon markets, including the EU ETS, will move towards greater integration. The EU ETS is likely to have a pivotal role in the perspective processes of carbon market integration. Four factors underpin this expectation: a) the size of the EU ETS, which remains today the world's largest ETS in operation; b) the number of countries participating in the EU ETS, which implies a strong track record in negotiating different interests; c) the experience accumulated by the EU in managing the system through various regulatory challenges; and d) the will of the EU to keep a leading role in global climate action.

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Appendix

A.1 Article 6 of the Paris Agreement

1. Parties recognize that some Parties choose to pursue voluntary cooperation in the implementation of their nationally determined contributions to allow for higher ambition in their mitigation and adaptation actions and to promote sustainable development and environmental integrity.

2. Parties shall, where engaging on a voluntary basis in cooperative approaches that involve the use of internationally transferred mitigation outcomes towards nationally determined contributions, promote sustainable development and ensure environmental integrity and transparency, including in governance, and shall apply robust accounting to ensure, inter alia, the avoidance of double counting, consistent with guidance adopted by the Conference of the Parties serving as the meeting of the Parties to this Agreement.

3. The use of internationally transferred mitigation outcomes to achieve nationally determined contributions under this Agreement shall be voluntary and authorized by participating Parties.

4. A mechanism to contribute to the mitigation of greenhouse gas emissions and support sustainable development is hereby established under the authority and guidance of the Conference of the Parties serving as the meeting of the Parties to this Agreement for use by Parties on a voluntary basis. It shall be supervised by a body designated by the Conference of the Parties serving as the meeting of the Parties to this Agreement, and shall aim:

(a) To promote the mitigation of greenhouse gas emissions while fostering sustainable development;

(b) To incentivize and facilitate participation in the mitigation of greenhouse gas emissions by public and private entities authorized by a Party;

(c) To contribute to the reduction of emission levels in the host Party, which will benefit from mitigation activities resulting in emission reductions that can also be used by another Party to fulfil its nationally determined contribution; and

(d) To deliver an overall mitigation in global emissions.

5. Emission reductions resulting from the mechanism referred to in paragraph 4 of this Article shall not be used to demonstrate achievement of the host Party's nationally determined contribution if used by another Party to demonstrate achievement of its nationally determined contribution.

6. The Conference of the Parties serving as the meeting of the Parties to this Agreement shall ensure that a share of the proceeds from activities under the mechanism referred to in paragraph 4 of this Article is used to cover administrative expenses as well as to assist developing country Parties that are particularly vulnerable to the adverse effects of climate change to meet the costs of adaptation.

7. The Conference of the Parties serving as the meeting of the Parties to this Agreement shall adopt rules, modalities and procedures for the mechanism referred to in paragraph 4 of this Article at its first session.

8. Parties recognize the importance of integrated, holistic and balanced non-market approaches being available to Parties to assist in the implementation of their nationally determined

contributions, in the context of sustainable development and poverty eradication, in a coordinated and effective manner, including through, inter alia, mitigation, adaptation, finance, technology transfer and capacity-building, as appropriate. These approaches shall aim to:

(a) Promote mitigation and adaptation ambition;

(b) Enhance public and private sector participation in the implementation of nationally determined contributions; and

(c) Enable opportunities for coordination across instruments and relevant institutional arrangements.

9. A framework for non-market approaches to sustainable development is hereby defined to promote the non-market approaches referred to in paragraph 8 of this Article.

A.2 Types of ETS linkage

Different forms of ETS linking can be classified into three categories (Lazarus *et al.*, 2015; Borghesi *et al.*, 2016; Schneider *et al.*, 2017; Quemin and de Perthuis, 2017):

- *Unilateral/Bilateral/multilateral*

  Unilateral linking is a one-way form of linking where entities from one ETS can purchase and use allowances from another ETS for compliance, but not *vice versa*[134]. Bilateral linking implies mutual recognition of units between ETSs, while multilateral linking involves a similar recognition across multiple ETSs.

- *Direct/indirect*

  Direct linking allows regulated entities in the linked ETSs to purchase and use each other's allowances for compliance with their domestic emissions reduction obligations. Indirect linking refers to a situation in which two programmes both recognise a unit from a third system.

- *Full/restricted*

  Full linking involves the unconditional mutual recognition of units without any quantitative or qualitative restriction. Restricted linking involves the partial, conditional or restricted

[134] Unilateral linking can also be considered as a restricted linking method, i.e. one jurisdiction limits the import of emissions allowances to zero while the other jurisdiction allows unlimited imports (Lazarus *et al.*, 2015).

recognition of units from other ETSs. Units used in each ETS are recognised as being valid in terms of compliance among the linked programmes but with some restrictions.

Focusing on restricted linking, four different methods have been proposed to date. They are quantitative restrictions, border taxes on allowances transfer, exchange rates, and discount rates:

- *Quantitative restrictions (or quotas)*

  It restricts the number or the type of units from other jurisdictions that can be used for compliance. Restrictions can be formulated and implemented in different ways on either side. They would be typically designed to limit overall net imports, rather than exports, and could be expressed as a fraction of the total compliance obligations that an entity might surrender.

- *Border taxes on allowances transfer*

  A border tax occurs when there is a cross-jurisdiction allowances transfer. Due to differences in domestic taxation systems and depending on whether an emissions allowance is classified as good or as service, multiple tax rates can be applied to traded emissions allowances. Border taxes can be imposed by either the exporting jurisdiction or, more typically, the importing jurisdiction.

- *Exchange rates (or trading ratios)*

  The exchange rate adjusts the value of units transferred between jurisdictions. Units from one jurisdiction can be used for compliance in another, but their value is adjusted by applying a conversion factor. If the exchange rate between allowances of jurisdiction A and of jurisdiction B is set at, say, two, then entities from jurisdiction A can buy one unit allowance from jurisdiction B and use it as two units in jurisdiction A. Symmetrically, in this case two units of allowances imported from jurisdiction A is worth one unit of allowance in jurisdiction B.

- *Discount rates*

  Discount rates can be regarded as a variation on exchange rates since they also involve a conversion factor. However, they do not need to be set as reverse ratios between two linked systems. Typically, they can both be greater than one – more than one unit from another jurisdiction is required to meet internal compliance obligations. For example, both jurisdictions can apply a 20% discount to the value of allowances imported from the other jurisdiction. If this is the case, 1.25 imported allowances are required *per* ton emitted in the home jurisdiction.

Chapter 6

A multi-criteria evaluation of the EU ETS

1. Introduction

The EU ETS is the largest cap-and-trade scheme in terms of countries involved and emissions covered, and can be considered as the cornerstone of EU climate policy. Its policy evaluation is, therefore, key not only to achieving EU emission targets in a cost-effective manner, but also to guiding similar initiatives worldwide.

This chapter aims at performing a systematic evaluation of the EU ETS as a whole, based on relevant *ex-post* literature review on the EU ETS investigated from the perspective of five basic evaluation criteria proposed by the European Commission. Indeed, as stated by the European Commission (European Commission, 2013, 2017a) all EU policy interventions, such as the EU ETS, should be assessed according to five minimum evaluation criteria: (i) effectiveness, (ii) efficiency, (iii) coherence, (iv) EU added value and (v) relevance. The EU also acknowledges other criteria for EU policy assessment, namely: intertemporal sustainability, acceptability, utility, and complementarity with other Member States (MS) policies.

As suggested by the EC Guidelines on evaluation (European Commission, 2017a), in order to better investigate the criteria set by the EC, we propose a set of framing questions that can guide the evaluation of the specific criteria. It should be acknowledged, however, that answering some of these questions may address multiple criteria. Indeed, some issues may be discussed under more than one criterion and hence there may be some natural overlap. This is particularly true for the two criteria of effectiveness and efficiency, as both are at the very core of the cap-and-trade philosophy and are strongly interlinked through, for example, the issue of competitiveness. Potential overlap also exists between the relevance criterion and the EU-added value criterion, which assesses whether the EU ETS continues to be justified at the EU level.

Table 1 - The EU criteria for policy evaluation and relevant framing questions.

Criteria	**Framing questions**
Relevance	How well do the objectives of the EU ETS correspond to the needs of society? Is the EU ETS relevant for the international agenda? How relevant is the EU ETS to EU citizens?
Effectiveness	Has the EU ETS reduced GHG emissions? Has the EU ETS reduced GHG emissions cost-effectively?
Efficiency	Are the costs associated with the EU ETS proportionate to the benefits it has generated?
Coherence	Is the EU ETS coherent with other policies having similar objectives? Is the EU ETS coherent with wider EU policies? How well does the EU ETS complement other climate-related EU Directives (RES and EE)? And/or, how well do EU climate related policies (RES and EE) complement the EU ETS? Is the EU ETS coherent with international obligations?
EU added value	What is the additional value resulting from the EU ETS, compared to what could have been expected from MS acting at national and/or regional levels?

The rest of the chapter is organized according to the five evaluation criteria. For each of them we provide a description of the valuation criteria, the framework questions addressed and the relevant literature review. For reasons of consistency we will begin the evaluation starting from the relevance criterion, since its fulfilment is regarded as the *sine qua non* for the assessment of the other criteria.

2. Relevance

The relevance evaluation refers to the correspondence between the needs of society and policy objectives. As far as the needs of the European society are concerned, there is almost universal scientific consensus on the necessity of reducing and eventually eliminating anthropogenic GHG emissions. In this regard, the EU ETS has a twofold objective, namely to abate carbon emissions (in a cost-effective way) and to promote low carbon investments. These objectives are at the basis of the effectiveness criterion, which will be analyzed later on in this chapter. The relevance evaluation assesses the extent to which the EU ETS objectives correspond to the needs within the EU society (see the first question in the table above), and hence touches on aspects of policy design and on time consistency. The relevance evaluation criterion assesses the mismatch between policy objectives and needs taking into account the dynamic nature of the needs and the existence of changing circumstances. Certain objectives may be met or superseded, while new needs and problems may arise or change.

Relevance evaluation drives policy-makers' decisions whether continuing, changing or stopping a policy intervention. Relevance analysis is crucial because if an intervention does not address current needs then it should not be pursued, no matter how effective, efficient or coherent it may be. In other words, if a policy or intervention is not relevant all other evaluation criteria will no longer be appropriate. For this reason, there exists a strong link between relevance evaluation and the EU added value criterion, which assesses whether action continues to be justified at the EU level compared to what could have been expected from MS acting at national and/or regional levels.

To investigate whether and to what extent the EU ETS is still relevant, it can be claimed that an efficient and effective EU ETS system is needed more than ever today in order to mitigate current negative climate change effects, and to comply with international and regional targets and commitments on carbon emission reductions. There is urgency to reduce carbon emissions, which explains why the EU ETS was progressively extended to further sectors. Indeed, from phase III the sectoral scope was expanded to include other sectors, namely aluminium, carbon capture and storage, petrochemicals and other chemicals. A notable exception is the aviation sector which was partially included in the ETS sectors only in 2012: the EU ETS covers only emissions from flights within the European Economic Area (EEA), pending the adoption of a regulation by the International Civil Aviation Organization (ICAO) on all international flights.

As to the relevance of the EU ETS in the context of the international agenda (the second question in Table 1), the EU ETS can certainly contribute to achieve the goals of the Paris Agreement, both within Europe and elsewhere through international cooperation. Indeed, article 6 of the Paris Agreement encourages voluntary cooperation between countries with carbon pricing mechanisms in place (Marcu, 2016).[135] In this regard, the possibility of linking the EU ETS with other cap-and-trade systems outside the EU could further enhance the relevance of the EU ETS, contributing to the progressive extension of carbon markets around the world.

As to the EU ETS relevance to EU citizens (the last question in the table), one can identify many different ways in which the EU ETS may be relevant for the European society. Among them, the most prominent one is probably its impact on human health. The link between air quality and human health has been widely acknowledged (see, among the others, European Environment Agency, 2017). The EU ETS covers only specific GHGs (i.e. CO2, N2O, PFCs); however, the sectors regulated by the EU ETS produce also local pollutants (e.g. SO2, NOX, CO, PM10), which are responsible for large impacts on human health. As Wagner and De Preux (2016) have pointed out, therefore, by reducing the overall emissions of the regulated sectors, the EU ETS can produce remarkable co-benefits on human health. Such co-benefits, however, are difficult to measure, and further analyses will be needed in the future to get a deeper understanding on the specific impacts of the EU ETS on human health.

3. Effectiveness

The effectiveness criterion for policy evaluation refers to the success of a policy in meeting its objective. The objective of the EU ETS, as stated in the ETS Directive, is "to promote reductions of greenhouse gas emissions in a cost-effective and economically efficient manner" (European Parliament and Council, 2003, 2009a). In other words, the mission of the EU ETS is to cut emissions and to do it at minimum cost ("cost-effective"), while also being economically efficient. Cost effectiveness in reducing emissions is the main rationale for emission trading and more generally for carbon pricing. It is the reason why economists tend to favour carbon pricing over command-and-control approaches (e.g., emission efficiency

[135] Further analysis on Article 6 of the Paris Agreement will be conducted through the chapter under the coherence evaluation criteria.

standards) to climate mitigation. The additional requisite of economic efficiency ("economically efficient manner") further specifies that the cost of abating emissions needs to be proportionate to the benefit that the same action produces.

In this section, we discuss whether the EU ETS has reduced emissions and whether it has done it cost-effectively. The question of economic efficiency is addressed in the next section since in the European Commission's framework for policy evaluation (economic) efficiency is a separate criterion, distinct from effectiveness and the others.

3.1 Has the EU ETS reduced GHG emissions?

Taken together, the relevant empirical literature indicates that the EU ETS has contributed to the emission reductions of regulated agents. The same literature, however, does not offer exhaustive, nor even highly consistent evidence on the magnitude of emission reductions attributed to the EU ETS. Firstly, econometric estimates of emission abatement are limited to Phases I (mainly) and II. Hence, more up-to-date studies are desirable. Secondly, the magnitude of estimated effects is quite variable. This variability may be at least partly explained by the generality of the effects; that is, by whether they are average effects referring to multiple countries and sectors or, viceversa, they refer to specific countries or sectors. The former tend to find emission reductions in the order of 2%-3% of business-as-usual (BAU) emissions, i.e. counterfactual emissions had the EU ETS not been in place (e.g., Ellerman and Buchner, 2008; Ellerman *et al.*, 2010; Anderson and Di Maria, 2011; Meyer and Meyer, 2013; Gloaguen and Alberola, 2013). By contrast, some of the latter find much more substantial emission reductions, in the order of 15%-25% of BAU emissions (e.g., Petrick and Wagner, 2014; Wagner *et al.*, 2014; Klemetsen *et al.*, 2016). A complicating factor for correctly interpreting this apparent discrepancy is that it might also be related to methodological differences. Indeed, the large effects are all found using the difference-in-differences (DiD) approach (which mimics a randomised control experiment), while the studies deriving BAU emissions based on aggregate data of regulated sectors all find more modest effects (Table 2).

Table 2 – Surveyed literature relevant to emission abatement.

Study	Phase	Year	Sectors	Methodology	Main results
Ellerman and Buchner (2008)	1	2005-2006	All	BAU calculated with aggregated data on emissions/GDP.	3% reduction due to the EU ETS

Ellerman *et al.* (2010)	1	2005-2007	All	BAU calculated with aggregated data on emissions/GDP.	Abatement of 210 [120-300] Mton (3.5%). EU15 accounted for most of the abatement and Germany accounted for 40% of the EU25 abatement.
Anderson and di Maria (2011)	1	2005-2007	All	BAU calculated with cross-country dynamic panel data (2000-04) and compared with verified EUAs 2005-07.	Total abatement of 175 MtCO2 (-2.8%).
Meyer and Meyer (2013)	1	2005-2008	All	BAU calculated with a dynamic input-output simulation (GINFORS).	Total abatement of 1 to 3% compared to counterfactual scenario without ETS.
Egenhofer *et al.* (2011)	1, 2	2005-2009	All	BAU calculated with aggregated data on emissions/GDP.	Improvement of the energy intensities (compared to BAU scenario). Stronger in phase II.
Gloaguen and Alberola (2013)	1, 2	2005-2011	All	Cross-country panel data and comparison to BAU scenario	Of the whole emission reduction estimated, renewable energy accounted for 50-60% (600-700Mt), economic crisis for 20-30% (300Mt)
Abrell *et al.* (2011)	1, 2	2005-2009	All	Firm-level DiD	Emissions reduced by -3.6% in phase II compared to phase I.
Petrick and Wagner (2014)	1, 2	2007-2010	Manufacturing (Germany)	Firm-level DiD	Phase I had no effect. Phase II increased emission reduction by 25-28%.
Wagner *et al.* (2014)	1, 2	2000-2010	Manufacturing (France)	Firm-level DiD	No effect during phase I but significant reduction in phase II (13-20%).
Jaraite and di Maria (2016)	1	2004-2007	All (Lithuania)	Firm-level DiD	The ETS did not cause emission reduction over phase I.
Klemetsen *et al.* (2016)	1, 2, 3	2001-2013	Manufacturing (Norway)	Firm-level DiD	Phases I and III had no effect on emissions. Only phase II shows evidence of

					emission reduction (33%, significant at 10% level).

Thus, reconciling this apparent discrepancy between the two literature subsets, explaining what is due to methodological differences and what is due to idiosyncratic factors, is also a desirable objective for future research. Finally, despite said differences, a robust general result seems to emerge from the literature, indicating that emission abatement was more significant during Phase II than during the first trading period.

3.2 Has the EU ETS reduced GHG emissions cost-effectively?

Evaluating whether the EU ETS (or any ETS for that matter) has reduced emissions in a cost-effective way is a question that is best addressed by first distinguishing between the short term and the long term. Perhaps not sufficiently emphasised in the literature, this distinction bears important theoretical and practical implications. Moreover, as Fuss *et al.* (2018) argue, since the ultimate goal of an ETS is to achieve a long-term cap (rather than a series of annual caps), the dynamic long-term perspective can be considered more appropriate for assessing its performance. The static short-term perspective is also needed, however, most notably because it allows evaluating the market of emission allowances.

Short-term perspective

Under the static short-term perspective, the technology used by regulated firms is considered fixed (*i.e.*, technological changes are not considered). Accordingly, emissions are reduced at minimum total cost if marginal abatement costs are equal across regulated firms. That is, cutting an additional tonne of carbon costs the same for all regulated firms. With a carbon tax, this condition for cost-effectiveness is verified by design (as long as a single tax rate is applied). With an ETS, by contrast, marginal abatement costs need to be equalised through the market of emission allowances. In formal terms, $P = MAC_i \quad \forall i$, where P is the price of emission allowances and MAC_i is the marginal abatement cost of firm i. For this to happen, the market of emission allowances must be efficient, which means that the allowance price reflects all available, relevant information, implying that the gains from trade are fully realised. On the other hand, market efficiency requires that market power and transaction costs are not significant (Tietenberg and Johnstone, 2003).

As part of a broader investigation of price and market behaviour in the EU ETS, Hintermann *et al.* (2016) review the empirical literature relevant to market efficiency. The authors mention a few studies that infer evidence of market inefficiency, in the first half of Phase II, by identifying opportunities for systematic profitable trading (not possible under market efficiency). Concerning transaction costs specifically, another handful of empirical studies exist which look at Phase I or Phase II.[136] Taken together, they provide robust evidence that smaller regulated firms are less active in trading due to both lack of institutional capacity for optimal trading and higher transaction costs per tonne of emissions. Transaction costs in the EU ETS are mainly administrative in nature and may be explained by start-up costs, which include the costs of estimating baselines and measuring equipment, monitoring, reporting and verification (MRV) costs, as well as trading costs. Transaction costs are hard to quantify, with existing evidence mainly drawn from qualitative studies. Focusing on international offset credits, Naegele (2018) estimates foregone revenues from trading, in Phase II, to have been over €1.3 billion. While acknowledging that also other factors were most likely in play, the author relates this found evidence of market inefficiency to transaction costs. It is important to note that the reform for Phase III (European Parliament and Council, 2009) addressed the issue of excessive transaction costs by allowing small installations to opt-out of the system if they are subject to equivalent measures. However, to our knowledge, a reassessment of transaction costs after Phase II does not yet exist. Regarding the possibility of large regulated firms exercising market power, Hintermann (2017) appears to be the only existing empirical study applied to the EU ETS. Looking at ten electricity firms during Phase I, the author infers that some firms' excess allowance holdings were consistent with strategic price manipulation.

All in all, although some empirical studies have pointed to market inefficiencies, the general conclusion drawn by Hintermann *et al*. (2016) is that the EU ETS as a carbon market has so far performed quite well, noting that the dynamics of the allowance price has been clearly related to market fundamentals. Moreover, the EU ETS can be expected to become more efficient in the future. Indeed, compared to Phases I and II, a number of factors should result in greater market efficiency: extended auctioning and harmonised free allocation, the possibility to opt-out for small installations, learning-by-doing in allowance trading (on the part of regulated firms) and prospective higher allowance prices (favouring optimal abatement and trading decisions).

[136] See Naegele (2018) for an up-to-date summary of the literature.

Dynamic long-term perspective

Under the dynamic long-term perspective, the technology used by regulated firms is not fixed. Crucially, this implies that the price of emission allowances, by virtue of driving investment decisions, is a key determinant of long-term emissions abatement costs. Moreover, the absence of both transaction costs and market power is not a sufficient condition to ensure the long-term cost-effectiveness of an ETS. Fuss *et al.* (2018) identify three reasons why allowance prices may deviate from the optimal path (*i.e.*, the path that minimises the total cost of meeting the long-term cap): investors' myopia, investors' excessive discounting, and lack of credibility of the long-term cap. Estimates of optimal allowance price paths are produced by intertemporal energy system optimisation models, of which not many exist. The same authors report that the PRIMES model used by the European Commission estimated optimal carbon prices to be €25 per tonne (of CO_2) in 2020 and €50 in 2050. For several years, low carbon prices determined by excess supply of emission allowances (caused by the Great Recession, negative interactions with energy policies and the use of international offsets) have been the main issue of the EU ETS. In response to it, important measures have been taken, including notably the establishment of the Market Stability Reserve (MSR) and the further tightening of the future cap (as part of the Reform for Phase IV), which have proved to be effective. At the time of writing, December 2018, the price of emission allowances has been floating between €20 and €25 per tonne, which clearly is not far off from the €25 above. Thus, the EU ETS appears to have been put back on a path that, at present at least, is consistent with long-term cost-effectiveness.

As to the past, relevant empirical evidence concerns the impacts of the EU ETS on low-carbon technological change and on international competitiveness of regulated firms. Though evidence is still limited to the first two trading periods, our literature review indicates that the EU ETS mainly stimulated small-scale low-carbon investments resulting in incremental emission reductions. Importantly, however, the EU ETS also caused a substantial increase in low-carbon patents (Chapter 4). Generally modest carbon prices and generous free allocation also explain why, on the whole, no significant negative effects of the EU ETS on firms' competitiveness have been found (Chapter 2).

4. Efficiency

The efficiency criterion for policy evaluation refers to the benefit-cost balance of a policy: a policy is efficient in absolute terms if it achieves the highest possible benefit-cost ratio. In climate change economics, the two key metrics to gauge the efficiency of any mitigation policy are the cost and the benefit to society of reducing GHG emissions by an extra tonne of CO_2 (or CO_2-equivalent). They are called, respectively, the Marginal Cost of Abatement (MAC) and the Marginal Social Cost of Carbon (MSCC). In theory, for a policy to be efficient, it must reduce emissions up to the point the first matches the second.[137] However, this condition is not sufficient, as for the policy to be efficient it must also be cost-effective; that is, it must reduce emissions at least cost. Carbon pricing, whether in the form of carbon taxation or emission trading, is in principle cost-effective because it equalises marginal abatement costs across subject agents.[138]

In the EU ETS, under classical economic assumptions, the price of emission allowances is the level to which marginal abatement costs of subject operators converge through the market mechanism. Thus, the relevant comparison is between this price and the MSCC. Certainly in the last few years, the price of European emission allowances has been lower than most existing MSCC estimates, whose average is around \$30/tCO_2.[139] Though MSCC estimates should be taken as indicative rather than as punctual values,[140] this difference suggests that emission abatement has probably been suboptimal. In other words, there has been space for increasing social welfare by abating more emissions. As noted before, the recent reforms of the EU ETS, notably the establishment of the MSR and the tightening of the future cap, have gone precisely in this direction. As a result of these regulatory interventions, current prices of emission allowances clearly have got closer to a level that can be considered optimal also by the efficiency criterion.

5. Coherence

The coherence evaluation should provide evidence of where and how the EU ETS is achieving its objectives, if actions undertaken to achieve them are complementary or if they are potentially contradictory, and if there are approaches which are causing inefficiencies. The focus on coherence may vary depending on the type of evaluation and is particularly important in fitness checks, where coherence analysis will look for evidence of synergies or inconsistencies between policy objectives.

[137] For a given policy, if MAC < MSCC, social welfare can be increased by reducing emissions; if MAC > MSCC, social welfare can be increased by reducing emissions abatement.
[138] Again, for an ETS this is only true under certain assumptions. The discussion in Section 3 above explains why an ETS may in fact not be cost-effective, over the short- or the long-term.
[139] In Tol's (2011) review of existing MSCC estimates, the median MSCC is \$116/tC; that is, \$31.6/tCO_2.
[140] For a critical appraisal of MSCC estimates and of their possible use in climate policy, see van Den Bergh (2004) and Stern (2008).

The evaluation of coherence refers to how well the EU ETS fits together with other climate policies, both within the EU and internationally. For the EU policies, coherence evaluation is particularly relevant in relation to renewable energy (RES) and energy efficiency (EE) Directives, while regarding international climate policies, coherence evaluation should consider the interaction with international offset credits, international climate agreements and other carbon markets.

Since all these policies have an effect on EU carbon emissions, direct or sometimes indirect, they can affect the carbon price in way that could jeopardize the intended price signal and, subsequently, the induced cost-effectiveness pursued.

5.1 Internal coherence: between the EU ETS and other EU climate and energy policies

Coherence evaluation of the EU ETS may be interpreted as internal coherence with respect to other EU climate and energy policies, namely those set by the RES Directive (European Parliament and Council, 2009b) on the promotion of the use of energy from renewable sources and the Energy Efficiency Directive (European Parliament and Council, 2012). More specifically, in order to answer the first two framing questions reported in Table 1 for this criterion, it is worth noting that coherence can take on at least two different meanings. We can refer, first, with this term, to how the EU ETS and other policies are complementary with respect to static efficiency (emission reduction). But the term can refer, second, to dynamic efficiency (low-carbon technological change). It is important to emphasize that the explicit objective to incentivize low-carbon investment was set for the EU ETS after the RES and EE Directives were adopted (Healy, 2015). It is also worth noting that internal coherence analysis is made even more complex by the fact that both energy and environmental policies are shared competences in the EU legislative system (Articles 191 and 194 of the TFEU, Lisbon Treaty). This means that not only will other EU policies affect the EU ETS, but also that the ones conducted by individual MS will interact with it. This is particularly true for the RES Directive. Indeed, the latter has had a positive impact on the dynamic efficiency of the EU ETS by mandating MSs to increase their share of RES through the introduction of renewable energy support schemes at the national level. On the other hand, it is widely acknowledged that EU RES policies have contributed to lower the price of EUAs, thus reducing the dynamic efficiency of the EU ETS. According to some authors (Carraro, 2015), therefore, consistency with other EU climate policy measures (subsidy to renewables, in particular) is probably the most urgent issue to be tackled by the EU ETS reform.

When referring to static efficiency only, in *ex-ante* economic literature there is a broad consensus that an ETS only policy regime is preferable with respect to a broader policy mix. Several contributions, stressed as overlapping policies are likely to produce excess costs and no significant incremental emission reduction (see among others, Morris 2009; Böhringer *et al.,* 2009; Boeters and Koornneef, 2011). Nevertheless, predictions in this literature are completely different when dynamic efficiency and multiple market failures are considered. Fisher and Newell (2008), for instance, point out that the energy market is characterized by three main market failures: carbon emissions, learning spillovers and R&D spillovers. Therefore, a wider policy mix including emission prices, R&D subsidies and renewable energy subsidies would work best with respect to an ETS only policy portfolio. In the *ex-post* analyses on policy interaction, there seems, on the contrary, to be a much broader consensus on the topic. This literature highlights several different aspects that can provide a valuable answer for the third framing question "How well does the EU ETS complement the other climate-related EU Directives (RES and EE)?" and also for the complementary fourth question. Firstly, Weigt *et al*. (2013) and Van den Berg *et al*. (2013), show that there is a strong complementarity between RES deployment and the EU ETS, but that the impact of the two policies is different. In their counterfactual analysis, Weigt *et al*. (2013) estimate that, without RES, Germany's emission in the electricity sector would have been 11% to 20% higher with respect to the *status quo*. A similar result has been found when estimating the effect of the hypothetical absence of the EU ETS on emissions, holding RES constant, but the size of this effect is much smaller (1% to 3%). Secondly, one econometric study (Gloaguen and Alberola, 2013) assesses the importance of different factors, among which electricity generation and energy intensity, in explaining emissions from the sectors covered by the EU ETS, in the period 2005-2011. The authors calculate that a reduction of around 1.1 billion tCO2 have been achieved within the sectors covered by the EU ETS, and that more than half of this reduction resulted from RES and EE policies. Thirdly, Koch *et al*. (2014), in a study on the determinant of EUA prices estimate that about 25% of the variation in carbon prices can be attributed to RES-E generation, indicating the presence of an interaction effect –though modest– among the two policies. However, we notice that this literature is still rather scarce, and that more research effort is needed in this field.

Finally, it is worth noting here that the Market Stability Reserve (MSR) can play a relevant role in coping with possible negative interactions between the EU ETS and other policies. The MSR has the specific aim of coping with possible shock in allowance demands. In particular, the MSR introduces some degree of flexibility in EUA supply, in order to give an automatic response to all these external shocks, including RES deployment, which can generate allowances surplus.

5.2 External coherence: between EU ETS and international environmental policies

Answering the last framing question "To what extent is the EU ETS coherent with international obligations?" means talking about compliance with international agreements (the Paris Climate Agreement), EU interventions in developing countries (international offset credits), and/or with other carbon trade systems outside Europe (linking). The comparison points for coherence may vary according both to time and to the level of coherence.

As to the compliance with international agreements, the Paris Agreement (PA) represents a turning point in terms of general expectations with respect to EU ETS, although it was not as much in terms of efficiency (carbon prices are still very low) and effectiveness (low incentive for technology innovation investment). However, if we consider the coherence criterion the Paris Agreement has not changed in terms of emission reduction targets within the EU. In fact, in the Intended National Determined Contribution (INDC) the EU and its 28 MSs commit to a binding target of at least 40% domestic reduction in GHG emissions by 2030, compared to 1990. These reductions are to be fulfilled jointly, as set out in the conclusions by the European Council of 24 October 2014. But relevant differences exist between the EU INDC and the EU ETS, in terms of emissions covered and scope. EU INDC covers 100% of emissions (from energy, industrial processes and product use, agriculture and waste) *versus* around 45% of the EU ETS (which, instead, excludes not only agriculture and waste but also housing and transportation), and all GHG not controlled by the Montreal Protocol (CO2, CH4, N2O, HFCs, PFCs, SF6, NF3) *versus* only three GHG (CO2, N2O and PFCs) by the EU ETS.

As pointed out above, the most relevant aspect of the Paris Agreement and its interaction with the EU ETS is related to Article 6 (UNFCCC, 2015). This article encourages voluntary cooperation between countries with carbon pricing mechanisms and more specifically establishes a mechanism to produce "internationally transferred mitigation outcomes" (ITMOs), which can be used to fulfil the National Determined Contribution (NDC) of another Party to the UNFCCC Conference of the Parties (Marcu, 2016). The Paris Agreement does not intend to create a market or a price for carbon, but to provide the possibility of an international market should the Parties desire it on a voluntary basis. As opposed to the Kyoto Protocol (KP), article 6 of the PA refers to any mitigation outcomes, in as much as they are transferred internationally; while the KP articles are restricted by their reference to Assigned Amount

Units (AAUs), Emission Reduction Units (ERUs) (Article 6, KP), and Certified Emission Reductions (CERs) (Article 12, KP).

As for the coherence of the EU ETS with international offset credits, it was decided under the 2030 Framework that installations will no longer be able to use international credits for compliance, unless a sufficiently ambitious international agreement on climate change was achieved. The Paris Agreement, aiming to hold global average temperature increases to well below 2°C and to pursue efforts to achieve a limit of 1.5°C, advances key market mechanisms relating to carbon offsetting developed under the Kyoto Protocol. The CDM and Joint Implementation (JI) are project-based offsetting mechanisms linked to the KP and allowed by the Linking Directive in 2004 (European Parliament and Council, 2004). These offset credits not only lower the cost of ETS compliance for European industry, but also extend the price signal of the EU ETS to activities not originally covered by the scheme. The JI projects implemented within the EU might, also, have generated emission reductions from sectors not already covered by the EU ETS during the second phase of the EU ETS (Ellerman *et al.*, 2010). Nevertheless, both CDM and JI are expected to expire in 2020 with the end of the KP.

The Paris Agreement at Article 6 introduces the Sustainable Development Mechanism (SDM). The SDM is supposed to aim at reducing overall GHG emissions and not only at transferring it from one place to the other, as was the case with Clean Development Mechanism (CDM) under the Kyoto Protocol (Kachi, 2017). SDM also differs from the CDM because unlike this latter it can be used by any Party to the Paris Agreement towards its NDC (Marcu, 2016). However, the new SDM is not envisaged as a cap-and-trade mechanism and, thus, it will not be further considered here because it is not within the scope of the EU ETS.

On external coherence as related to other carbon trade systems outside Europe, the EC envisages the development of an international carbon market through the linking of domestic cap-and-trade systems. Linking of emission trading systems would allow more cost-effective greenhouse gas emission reductions. Linking the EU ETS with other cap-and-trade systems offers several potential benefits, including reducing the cost of cutting emissions across linked systems by increasing access to abatement opportunities, increasing market liquidity, stabilising the carbon price, levelling the international playing field and supporting global cooperation on climate change. EU legislation envisages linking through the mutual recognition of allowances between the EU ETS and any third country ETS, as outlined in Article 25 of the EU ETS Directive (European Parliament and Council, 2003). In November 2017, the EU signed an agreement with Switzerland to link their emission trading systems. The link will allow participants in the EU ETS to use allowances from the Swiss system for compliance, and *vice versa*. This is the first agreement of this kind for the EU and the first between two Parties to the Paris Agreement. Transparency from both partners is a pre-requisite for similar bilateral agreements to be achieved in the future. Indeed, transparency is essential to build up the mutual trust between jurisdictions that is needed for linking, and at the same time is a main pillar of the Paris Agreement, both substantively and procedurally, particularly in the framework of the global stocktake and of the requirement to communicate progress on NDCs on five-year intervals.

6. EU Added Value

In line with the EU principle of subsidiarity (Article 5.3, TEU, Lisbon Treaty)[141], EU added value criterion should consider whether undertaking EU policies provides additional value with respect to what would have resulted from national policies. The evaluation of EU added value brings together the findings of the other criteria, drawing conclusions based on the performance of the policy intervention and assessing whether it is still justified. EU added value may be the result of different factors such as coordination gains, legal certainty, greater effectiveness or efficiency, complementarities, etc.

The very nature of any emission trading system means that the larger the scheme the better its performance in terms of emissions reduced and cost effectiveness. In this regard, EU added value can be evaluated by assessing whether (and how much) a centralized ETS is more effective or efficient compared to a sum of national-based MS policies. In this sense, the main additional value of the EU ETS derives from the possibility to lower compliance costs by exploiting low-cost abatement opportunities through trade across all MS (Healy *et al*., 2015). Additional co-benefits may include improved levels of governance due to enhanced central coordination and legal certainty that can only be provided at the EU level.

There is a strong consensus that harmonised approaches have generally improved the implementation of the EU ETS: this is as opposed to decentralized EUA allocation mechanisms, which, instead, caused distortions due to different interpretations of the rules by single MS. To overcome such distortions, while phases I and II were characterized by National Allocation Plans (NAPs), from phase III a single EU-wide cap on emission allowances was introduced to replace the existing system of national caps. The approach towards harmonisation in the EU ETS also concerns the allocation of free allowances based upon benchmarking rules

[141] In areas in which the European Union does not have exclusive competence, the principle of subsidiarity defines the circumstances in which it is preferable for action to be taken by the Union, rather than the Member States.

that reward those installations with better emission performances. This issue touches the concept of equity among MSs, in the sense that the harmonisation of the allocation rules will reduce the differences in allocation levels across countries with similar carbon intensities. This would contribute to reducing the competitive distortions that were previously observed in phases I and II (Matthes *et al*., 2005).

A final observation can be made on how the Brexit referendum may affect the added value of the EU ETS. We have already observed how international cooperation can increase cost-effectiveness through linking. In the case of a hard Brexit there might be a delinking of the United Kingdom (UK) from the EU ETS with a decrease in the EU ETS's EU added value and cost-effectiveness. Indeed, the abatement cost is higher for the EU-27 than for the EU-28. Moreover, this problem might be worsened by the fact that the UK is the second-largest European emitter of GHG in Europe and plays a key role in the EU ETS, being a central node in the transaction network (Borghesi and Flori, 2018).

A different perspective on this issue is offered by Babonneau *et al*. (2017). The authors evaluate the potential economic impacts of Brexit on long-term EU climate policy, assuming a hard Brexit with a complete leave of the UK. They conclude that Brexit might have a significant negative impact on the UK's climate policy cost, while it would have a relatively positive effect on the EU-27. Indeed, their results show that in all examined scenarios (excluding the case of a full effort-sharing decision rule), the UK would suffer from an increase in the cost of its climate policy by leaving the EU, estimated to be around USD 43 billion. In contrast, the EU as whole could be better off and experience some welfare improvements, assuming that the UK has to implement its emission reductions through a domestic carbon price and that it is not allowed to participate in any EU ETS system.

Other studies are recently emerging in the literature on the potential implications of Brexit for both UK and EU (e.g. Hepburn and Teytelboym, 2017; Pollitt and Chyong, 2017; EU, 2017). Among them, Tol (2018) investigates the possible impact of a UK departure from the EU ETS and finds it to be limited on the remaining 27 member states, while being much larger for the UK in terms of increased compliance costs with its climate policy targets, transition costs to replace the EU ETS and possible business loss as the carbon trade leaves the UK.

Within the EU, however, we may expect winners and losers. The MSs that are net sellers of permits will likely suffer from revenue loss whereas net buyers may benefit from lower prices. Moreover, Babonneau *et al*. (2018) argue that the non-participation of the UK in the existing EU ETS would be to the disadvantage of energy-intensive industries. Finally, the authors point out that the exit of the UK from the EU ETS would also have side effects, reinforcing the leadership role of other EU countries, especially EU funding MSs, particularly Germany that is the largest emitter in Europe (UNFCCC, 2016).

While a delinking of the UK is a possibility that should be taken into account, at the moment of writing the situation is still very uncertain and open to many possible outcomes that will need further analyses in the future.

7. Conclusions

The EU ETS is now in its second decade and approaching the end of its third phase. It reached its maturity growing through the unavoidable difficulties experienced in its early learning phases and the unexpected recession that hit Europe and the world economy after 2008. As a consequence, the EU ETS needed to adjust to rapidly changing circumstances and further modifications may be expected to address new economic and climate challenges. When originally conceived, it was designed to comply with the Kyoto Protocol targets. Now the Kyoto Protocol has *de facto* come to an end, and in 2015 a new global climate agreement was adopted at COP21 in Paris. The Paris Agreement has radically changed the design and governance of global GHG emission reduction, thanks to an innovative bottom-up approach. The introduction of the Nationally Determined Contributions (NDC) has given the UNFCCC Parties and, therefore, also the EU MS, the possibility of voluntarily committing to GHG emission reduction targets through their own modalities and according to their own natural and financial resources. The Paris Agreement was key not only for its innovative design, but also because it generated very high expectations in terms of the reach of global emission reduction targets. Hence, the Paris Agreement will probably have important effects for the EU ETS in the longer term, and these will need to be taken into account in a future *ex-post* literature review.

To assess the performance of the EU ETS, the present literature review has been conducted adopting five evaluation criteria (relevance, efficiency, effectiveness, coherence, EU added value), all of which have been associated with specific review questions. Focusing on these criteria we can draw the following conclusions. As to relevance, the EU ETS was considered, from its beginnings, to be the main instrument to reduce GHG emissions in the EU for the energy and industrial sectors and it still plays the key role there. More than ever an efficient and effective EU ETS system is needed in order to mitigate negative climate change effects, and to comply with the Paris Agreement targets and the EU NDC commitments on carbon emission reductions, as well as with the Energy Union calling for decarbonising the EU economy (European Commission, 2015). The Paris Agreement features a voluntary stocktake of how national pledges are contributing to a long-term target in 2018 and a voluntary revisiting of pledges in 2020. However, in 2023 a first binding stocktake is foreseen. Therefore, from 2023, if not before, the EU is expected to communicate how their national pledges, including those concerning GHG abatement, have contributed to the Paris Agreement targets. In this framework an efficient and effective EU ETS is central if the European Union is to respect its commitments.

In terms of effectiveness, it can be argued that over-allocation and periodic price instability have reduced the EU ETS's effectiveness, with periods of low incentive for cutting emissions and for investing in low-carbon technologies. However, the reviewed *ex-post* literature suggests that the ETS did contribute to abating carbon emissions, though the same positive results are not evident with the ETS's capacity to foster innovation. Phase II appears to be the most effective for abatement, though renewable energies and the economic downturn have proved to be more relevant in terms of overall emission reduction. At the firm level recent studies show carbon price having a significant effect on regulated firms' emissions (Petrick and Wagner, 2014), as opposed to a trivial effect on companies' investments in clean technologies (Laing *et al.*, 2014). This abatement is visible in both the power and the industrial sectors. It is relevant because the mechanism through which these two broadly-defined sectors abate emissions is qualitatively different. For the industrial sector abatement entails structural changes in production systems, whereas for the power sector abatement occurs through fuel switching within the merit order mechanism. The power sector was thought to have undertaken most of the carbon reduction and this has actually been the case. However, changes in the carbon-intensity of production in the industrial sector were achieved by the EU ETS.

On efficiency, relating to competitiveness there is no evidence of detrimental impacts on EU regulated firms. However, most of the existing literature on this topic is not sector-specific, but looks, rather, at the average competitiveness effect of the EU ETS across different sectors. The lack of clear effects on competitiveness might reflect the existence of low carbon prices and also the free allocation of allowances that contributed to prevent carbon leakage. The observed allowances over-allocation and the consequent fall in carbon prices cannot be attributed solely to the 2008 economic downturn, but it was the result of multiple factors including global and internal developments, such as the use of international credits, the 2012 foreign debt crisis, and renewable energy growth. Over-allocation and price decline have been frequently observed also in other cap-and-trade schemes similar to the EU ETS. As for another aspect of efficiency, concerning cost pass-through and windfall profits, there is a general consensus in the reviewed literature that there has been a substantial pass through of carbon costs in the energy sector and that in phases I and II windfall profits were high, in particular for electricity producers. However, some evidence is emerging that the energy-intensive industrial sector has also benefited from windfall profits due to free allocation in phases I and II.

On coherence, it could be argued that over the short term, energy policies overlapping with the EU ETS increased the cost of reducing regulated emissions, unless their abatement costs were lower than allowance prices. Over the long term, meanwhile, energy policies overlapping with the EU ETS put downward pressure on carbon prices, by decreasing the demand for emission allowances. In this way energy policies weakened the economic incentives for low-carbon innovation and investment. Due to the empirical challenges that *ex-post* analyses entail, assessing the costs differences between alternative policy combinations is a difficult task, as it requires the creation of credible counterfactual scenarios. Interestingly, all *ex-post* studies deal specifically with the interactions between the EU ETS and policies supporting electricity from renewable energy sources.

On EU added value, the main result of this criterion evaluation is that the additional value of the EU ETS is to lower compliance costs by enlarging the market and facilitating the trade in low cost abatement opportunities across all MSs. In short, the more emissions covered the lower the abatement cost opportunities. There is consensus in the literature reviewed that harmonised approaches have generally improved the implementation of the EU ETS with respect to decentralized allowance allocation mechanisms. In particular, the new centralized benchmark-based allocation mechanism, as opposed to the previous national allocation plans (NAPs), helped prevent the over-allocation of allowances experienced in phases I and II, which was originally due to exceedingly generous national schemes. Finally, another important added value of the harmonisation of the allocation rules concerns its equity implications among MSs, as it helped to reduce the differences in allocation levels across countries with similar carbon intensities.

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