Impact of climate policies on energy-intensive industries

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Impact des politiques climatiques sur les industries énergie-intensives

Impact of climate policies on energy-intensive industries

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Directeur de thèse: Philippe Quirion Frédéric Branger

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Résumé

Cette thèse étudie les fuites de carbone et les pertes de compétitivité dans les industries énergie-intensives générées par des politiques climatiques inégales. Après une méta-analyse des études de modélisations évaluant les fuites de carbone en cas de politique climatique unilatérale avec ou sans Ajustements Carbone aux Frontières, nous utilisons de l'économétrie des séries temporelles pour établir l'absence de preuves de fuites de carbone opérationnelles liés à des pertes de compétitivité dues du Système Communautaire d'Echanges de Quotas d'Emissions (SCEQE) pour le ciment et l'acier. Ensuite, nous décomposons les émissions du secteur cimentier Européen en sept effets avec la méthode Log Mean Divisia Index, et montrons que les variations peuvent être attribuées principalement à l'effet d'activité. Les réductions d'émissions apportées par le SCEQE sont estimées à 2% entre 2005 et 2012 tandis que le montant des "profits de rente" est évalué à 3.5 milliards d'euros. D'autre part, nous démontrons que l'industrie cimentière a réagi stratégiquement à l'introduction d'une nouvelle règle censée diminuer le nombre d'allocations gratuites pour les installations en sous-production. Les entreprises ont augmenté artificiellement la production dans certaines usines, générant des distorsions allant à l'encontre de la transformation bas-carbone du secteur. Enfin, nous discutons de réformes possibles pour le SCEQE et plaidons pour des allocations proportionnelles à la production pour le court terme.

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Impact of climate policies on energy-intensive industries

Abstract

This thesis contributes to the literature on carbon leakage and competitiveness losses in energy-intensive industries generated by uneven climate policies. After a meta-analysis of modelling studies assessing carbon leakage with or without Border Carbon Adjustments; we use time series econometrics and find no evidence of competitiveness-driven operational leakage due to the European Union Emissions Trading System (EU ETS) for steel and cement. Next, we decompose emissions the European cement sector into seven effects with a Log Mean Divisia Index method, and show that most of the variations can be attributed to the activity effect. Abatement due to the EU ETS is estimated at around 2% between 2005 and 2012 while the amount of "overallocation profits" is assessed at 3.5 billion euros. Further, we demonstrate that the cement industry strategically reacted to the introduction of a new rule supposed to reduce free allocation in low-producing installations. Companies artificially increased production in some plants, generating distortions going against the low carbon transformation in this sector. We finally discuss possible reforms in the EU ETS and advocate for output-based allocation in the short term.

Mots-Clés

Fuites de Carbone Compétitivité SCEQE Marché de permis d'émissions Industries lourdes Politiques climatiques Ajustements carbone aux frontières Méta-analyse Méta-régression Industrie cimentière ARIMA Prais-Winsten LMDI Abatement Surallocation Efficacité énergétique Seuil de niveau d'activité

Keywords

Carbon leakage Competitiveness EU ETS Emissions Trading EITE industries Climate Policy Border Carbon Adjustments Meta-analysis Meta-regression analysis Cement industry ARIMA Prais-Winsten LMDI Abatement Overallocation Energy efficiency Activity Level Threshold

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Chapter 3: Branger, F., P. Quirion and J. Chevallier (Forthcoming). Carbon leakage and competitiveness of cement and steel industries under the EU ETS: much ado about nothing. *The Energy Journal*.

Chapter 4: Branger, F., P. Quirion (2015). Reaping the carbon rent: Abatement and overallocation profits in the European cement industry, insights from an LMDI decomposition analysis. *Energy Economics* 47, 189-205.

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Acronyms

ADF	Augmented Dickey-Fuller
ALCF	Activity Level Correction Factor
ALTs	Activity Level Thresholds
ARE	Abatement Resource Effect
ARIMA	AutoRegressive Integrated Moving Average
BAT	Best Available Technology
BCAs	Border Carbon Adjustments
BOF	Basic Oxygen Furnace
CCS	Carbon Capture and Storage
CDM	Clean Development Mechanism
CER	Certified Emission Reduction
CGE	Computable General Equilibrium
CSCF	Cross Sectoral Correction Factor
CSI	Cement Sustainability Initiative
CSR	Corporate Social Responsibility
EAF	Electric Arc Furnace
EITE	Energy-Intensive Trade Exposed
ERU	Emission Reduction Unit
ETR	Environmental Tax Reform
EUA	European Union Allowance
EU ETS	European Union Emissions Trading System
EUTL	European Union Transaction Log
GATT	General Agreement on Tariffs and Trade
GDP	Gross Domestic Product
GHG	Greenhouse Gases
GNR	Getting the Numbers Right
GTAP	Global Trade Analysis Project
HAL	Historical Activity Level

HEL	Historical Emissions Level
ICAO	International Civil Aviation Organisation
LD	Low Demand
INDC	Intended National Determined Contribution
LMDI	Log-Mean Divisia Index
MD	Moderate Demand
MFN	Most Favoured Nation
Mt	Million tons
NT	National Treatment
OBA	Output-Based Allocation
OPEC	Organization of Petroleum Exporting Countries
PE	Partial equilibrium
PPM	Product and Production Method
PV	Photovoltaics
REML	Random Effect Multi-Level
SCM	Subsidies and Countervailing Measures
ULCOS	Ultra Low Carbon Steel Making
UNEP	United Nations Environmental Program
UNFCCC	United Nations Framework Convention on Climate Change
WBCSD	World Business Council on Sustainable Development
WTO	World Trade Organization

Introduction

International negotiations have failed to achieve what standard economic theory recommended: imposing a globally uniform price on greenhouse gas emissions that approximate their social costs (Tirole, 2009), thereby achieving climate change mitigation in the most cost-effective way. Paris agreements in 2015 are expected to shape the post-2020 global climate architecture in a bottom-up approach, each country submitting an Intended National Determined Contribution[1] or INDC, including goals and policies to achieve them. Climate policies have been and will continue to be uneven across the world, setting a challenge to move forward a low-carbon economy.

Indeed, one of the main reasons of the worldwide lack of ambitious climate policies is the possible threat to the competitiveness of heavy industries and the resulting carbon leakage. The asymmetry of carbon costs between regions may induce immediate losses of market share to the benefit of foreign competitors (operational leakage) or in the longer run location of energy-intensive industries in regions with more favorable climate policies (investment leakage)(Reinaud, 2008). As a result, emissions would rise in non-constrained countries, causing so-called carbon leakage. As carbon dioxide is a global pollutant (i.e. the geographic location of emissions has no influence on its environmental impacts) it would weaken or nullify climate policy efficiency. Moreover, the additional cost generated by climate policies may reduce the domestic industry's market share, destroy jobs and reduce profits. Such adverse effects are grouped together under the heading of a

[1] What is included in INDC is at the discretion of the States. It may include short or long term targets in terms of emissions or renewable capacites, policies to achieve these targets and other elements like need for international support or intended provision of finance (Ecofys, 2014).

loss in "competitiveness," a term the popularity of which is inversely proportional to its clarity.

The risk of carbon leakage is not uniformally distributed across sectors, but is complex to assess as it depends on various mechanisms (Ecofys and Oko Institut, 2013). First, the sector has to face a high carbon cost, which depends on its carbon intensity and its abatement potential. Second, imports and exports must be sensitive to domestic price, which depends on international trade barriers, bargaining power of the sector, products homogeneity and transportability. Finally, there must not be relevant substitutes with low carbon intensity. Because exports of electricity are very limited, the power sector is not exposed to carbon leakage, but it is different for the so-called energy-intensive trade exposed (EITE) sectors which include cement, iron and steel, aluminium, chemicals, pulp and paper, oil refining, glass or ceramics.

Political debates essentially focus on negative outcomes of unilateral climate policies related to global emissions and competitiveness, but positive aspects, mostly due to technological change and innovation, can also be highlighted. Porter and Van der Linde (1995) contested the established paradigm that strict environmental regulation was necessarily harmful for business, and claimed that "properly crafted" regulation could enhance competitiveness through innovation. Jaffe and Palmer (1997) further distinguished the "weak" version (properly designed environmental regulation fosters innovation) from the "strong" version (benefits of innovation can outweigh compliance costs so the overall impact on competitiveness is positive) of the Porter Hypothesis. After twenty years of studies, a general consensus prevails to support the "weak" version while results are more mixed for the "strong" version (Ambec et al., 2013). Pioneering climate policies can also encourage the development of new eco-industries. Domestic firms could then benefit from a First Mover Advantage (Lieberman and Montgomery, 1988) and seize export opportunities in new emerging markets. For example, the world leader in the wind power industry is the Danish company Vestas, which benefitted in its early stages of development from the ambitious national wind policy. At the same time, the competitive advantage of climate pioneers is not guaranteed, and dominant positions in industries featuring fast technical progress are fragile. Finally, the diffusion of low carbon technologies outside the carbon constrained region, called climate spillovers (Golombek and Hoel, 2004), generates negative leakage. Renewable installed capacities in China (which has become the world leader in wind installed capacities since 2010[2] and is soon to become the leader in photovoltaics installed capacities) would certainly not be as high had feed-in tariffs not been implemented in Europe and in the United States.

Because of the EITE industries' influence over the policy process, specific measures to protect these sectors are systematically integrated in climate policy packages. Broadly defined, the main two options are Border Carbon Adjustments (BCAs), which consist in reducing the carbon price differentials of the goods traded between countries, and free allocation of allowances or tax exemptions. More specifically, free allocation can be based on *ex ante* rules only (being computed with historical emissions, or historical production multiplied by a benchmark) or feature *ex post* adjustments (based on actual production, such as output-based allocation (OBA)). BCAs, which have been intensively discussed but never been implemented, will be discussed in Chapters 1 and 2. OBA, which have been implemented in the California carbon market since 2012, will serve as reference in Chapters 4 and 5.

In the major part of this thesis, the impact of climate policies on energy-intensive industries will be tackled through the example of the cement sector in the European Union Emissions Trading System (EU ETS). The unique age and size of the EU ETS, implemented in 2005, offers a natural field experiment. In addition, as the cement sector is highly carbon-intensive, the issues of carbon leakage, competiveness and free allocation rules are magnified. Before detailing the plan of the thesis, we will then provide some insights on the EU ETS and on the cement sector.

The cement sector represents 5% of global anthropogenic emissions and about 9% of emissions covered by the EU ETS. After reaching a peak in 2008 with 260 Mt, European cement production was severely impacted by the economic recession and never recovered (we estimate the production in 2013 at 180 Mt). Cement manufacture can be divided into two main steps: clinker production (90% to 95% of emissions, virtually all from direct emissions), and blending and grinding clinker with other material to produce cement (indirect emissions due to electric-

[2] http://www.gwec.net/global-figures/interactive-map/

ity use). The two broad options to decrease cement carbon intensity are then to either decrease clinker carbon intensity or reduce the clinker-to-cement ratio. Indeed, clinker substitutes such as blast furnace slag, fly ash or pozzolanas (volcanic rocks), being by-product of other industries or natural resources, have a much lower carbon intensity than clinker. Options to reduce clinker carbon intensity are limited by the fact that two third of emissions are process emissions that cannot be reduced. Only breakthrough technologies such as carbon capture and sequestration (CCS), or innovative cements not based on clinker, could significantly lower cement carbon intensity in the long run. Cement is very carbon-intensive and is a homogenous product. However, the industry presents other characteristics potentially limiting the risk of carbon leakage. It is poorly traded internationally, because of its bulkiness and because the raw material, limestone, is present virtually everywhere in the world. Besides, the cement sector is very oligopolistic and has regularly faced sanctions from national competition authorities. Producers have relatively high market power, especially inland compared to near the coasts because of transportation costs.

The EU ETS, presented as the "flagship" or "cornerstone" of the European climate policy, has celebrated its tenth anniversary in 2015. It covers approximately 45% of European emissions with a global cap of around 2 $GtCO_2$ spread across approximately 11,000 installations.[3] Beside domestic aviation, added in 2012, the covered sectors can be broadly distinguished between power and heat on the one hand, and manufacturing sectors on the other hand. The functioning of a cap-and-trade system goes as follows. First, allowances are auctioned or distributed to installations, the total amount of them being fixed by a global cap. Then, entities can buy or sell allowances to cover their emissions, and a carbon price emerges through supply and demand. The flexibility that trading brings is supposed to ensure that emissions are cut in the most cost-effective way. Direct sale of allowances by auctioning has significant economic advantages over free allocation (Hepburn et al., 2006). For example, the value of the auction revenues can be used to reduce other distortionary taxes and improve macroeconomic efficiency. Meanwhile, auctioning avoids windfall profits to polluters, incumbent and new firms are treated on an equal basis, and many other distortions that arise with free allo-

[3] http://ec.europa.eu/clima/policies/ets/index_en.htm

cation are avoided. However, in a world of uneven climate policies, free allocation is seen as necessary to mitigate carbon leakage.

After a first test phase (2005-2007) and a second phase corresponding to the period of compliance to the Kyoto protocol (2008-2012), the third phase started in 2013 and will last until 2020. Each phase is characterized by different rules, the most critical being whether allowances are auctioned or freely allocated to installations, and in the latter case the way they are distributed. Rules of the fourth phase (2020-2030) are currently under discussion, while those of the third phase will hardly be modified. For phase III, a list of sectors "deemed to be exposed to carbon leakage" was established using two quantitative indicators: the share of carbon costs relative to value added (using a CO_2 price of 30 euros per ton) and the trade intensity (imports plus exports in value divided by market size). These sectors, which represent the overwhelming majority of manufacturing emissions, receive free allocation based on historical production multiplied by sectoral benchmarks.[4]

The EU ETS has been the largest implemented carbon market and the worldwide reference in terms of cap-and-trade policy, setting an example for other carbon pricing initiatives (World Bank & Ecofys, 2014). However, this leadership position is increasingly being challenged. First, China should surpass the EU in terms of carbon market size in the following years, as the largest emitter on the planet will most likely implement a nationwide emissions trading system (Zhang, 2015). Second, a severe crisis and the inability of European authorities to engage meaningful structural reforms have undermined the system (Branger et al., 2015). A massive surplus of allowances, corresponding to more than a year of emissions, combined with the uncertainty in the stringency of long term targets, have driven down the carbon price. The latter has stayed below eight euros since 2012, compared to around 25 euros in mid-2008. The surplus of allowances is due to a combination of factors (Koch et al., 2014): the economic downturn, the effectiveness of renewable energy policies and the inflow of a significant number of cheap international credits. However, concerns about carbon leakage and competitiveness have aggravated the price drop by: (i) first reducing the environmental ambition

[4] In addition, a cross-sectoral correction factor (CSCF) is applied, equal to 0.9427, then declining at 1.74% per year. It ensures that the total amount of free allocation relative to the cap does not overcome a certain value. Finally, the activity level threshold (ALTs) rules reduce free allocation in case of important activity reduction (see Chapter 5).

of the EU ETS, and (ii) delaying market reforms intended to strengthen the carbon price signal.

We now turn to the plan of this thesis, which is structured around five independent chapters, corresponding to five published articles. Except for Chapters 1 and 2 which are advanced literature reviews essentially on *ex ante* studies, we adopt an *ex post* empirical approach in this thesis. Slight cuts have been made in the original articles in order to avoid redundancies, as well as minor updates related to policy issues and references.

Chapter 1 provides an in-depth literature review on carbon leakage and issues of competitiveness linked to uneven climate policies, and can be seen as an extension of this introduction. After a definition of key concepts, studies assessing the risk of carbon leakage are reviewed. We point out a discrepancy between *ex ante* (mostly CGEs models) and *ex post* studies (one of which being the chapter 3). Ex ante studies forecast a carbon leakage ratio (which corresponds to the emissions increase in the rest of the world related to home abatement) of 5-25% while *ex post* studies find no empirical evidence of carbon leakage. Policy packages addressing leakage and competitiveness issues are discussed, with an emphasis on BCAs. Models show that BCAs restore competitiveness of domestic energy-intensive industries and partially reduce carbon leakage, but have important distributional effects, shifting a part of the burden from the abating coalition to third countries. To implement BCAs, a series of technical choices have to be made, such as sectoral coverage, geographical coverage, inclusion of indirect emissions or exports, carbon content, or use of revenues. These choices would be crucial for the compatibility of BCAs with the World Trade Organization (WTO), which remain highly contentious among legal experts. Two past events, the WTO Shrimp-Turtle dispute and the attempt of inclusion of aviation in the EU ETS, shed some light on possible outcomes of unilateral imposition of BCAs. We argue that factors of success for their implementation include in-depth discussions with third countries on BCAs features prior to their implementation, flexibility provided to impacted countries to achieve comparable policies, relatively conservative values of carbon content and the altruistic use of revenues (handed back or routed to the Green Climate Fund).

Chapter 2 narrows down the literature survey with a more quantitative perspective, by investigating *ex ante* studies in energy-economy modelling assessing carbon leakage and competitiveness losses in energy-intensive industries. Following best practice guidelines of Nelson and Kennedy (2009), the meta-analysis aggregates 25 studies dating from 2004 to 2012 which included BCAs in one of their scenarios, altogether providing 310 estimates of carbon leakage ratios. First, descriptive statistics are presented. Kernel density estimations showing probability distributions of estimates are used to merge results across studies. A meta-regression analysis is then performed to explore the impact of different assumptions on the leakage ratio estimates, such as the size of the coalition, the abatement target, Armington elasticities (modelling international trade) and BCAs features. We find that, all other parameters being constant, BCAs reduce the leakage ratio by six percentage points. We also give statistical evidence that augmenting the size of the abating coalition reduces the leakage ratio while increasing the abatement target augments the leakage ratio. Finally, in the meta-regression, the inclusion of all sectors and the presence of export rebates appear to be the two most efficient features to reduce leakage, followed by the adjustment level based on foreign carbon content.

Following the previous quantitative survey of *ex ante* literature on carbon leakage, we contribute to the literature on *ex post* studies in Chapter 3 by investigating a potential competitiveness-driven operational carbon leakage due to the European Union Emissions Trading scheme (EU ETS). We focus on the two largest CO_2 emitters among European manufacturing sectors, cement and steel, and the first two phases of the EU ETS (2005 to 2012). From a simple analytical model, an equation is obtained linking net imports (imports minus exports) of cement and steel to local and foreign demand along with carbon price. The model implies that there should be a positive correlation between carbon price and net imports. The econometric estimation of this relation is made with two different econometric techniques, ARIMA and Prais-Winsten, which provide consistent results. Local and foreign demand are robust drivers of trade flows, but no significant effect of the carbon price on net imports of steel and cement is found. We conclude that there is no evidence of carbon leakage in these sectors, at least in the short run. These industries have benefitted from a large overallocation of allowances during this period, but free allowances bear an opportunity cost.[5] Therefore, provided one considers standard economic theory valid (companies behaving as profit-maximizers), the overallocation of allowances should not have had an influence on operational leakage. Our results would then suggest that within the historical price range for CO_2 (below 30 euros per ton), operational carbon leakage is not a serious threat for the energy-intensive industries.

Chapter 4 presents a detailed study on the European cement sector, made by cross-referencing the Getting the Numbers Right (GNR), database developped by the Cement Sustainability Initiative and the registry of the EU ETS, the European Union Transaction Log (where we collected plant-by-plant information on 276 cement plants). The variations of emissions from 1990 to 2012 are broken down using the Log-Mean Divisia Index (LMDI) method (Ang, 2004), both at the EU level and at the national level for six major producers (Germany, France, Spain, the United Kingdom, Italy and Poland). This method allows measuring the impact of seven effects on emission variations, which correspond to different mitigation levers that were previously discussed: activity, clinker trade, clinker share, alternative fuel use, thermal and electrical energy efficiency, and decarbonisation of electricity. We demonstrate that most of the emission changes in the EU 27 can be attributed to the activity effect though since the 1990s, there has been a slow trend of emission reductions mostly due to the clinker share effect (decrease in the clinker-to-cement ratio), the fuel mix effect and the electricity emission factor effect. Making assumptions on counterfactual scenarios, the abatement induced by the EU ETS is estimated at 2.2% $\pm$ 1.3% from 2005 to 2012, mostly because of a small acceleration in clinker reduction and alternative fuel use. However, we cannot exclude that these effects were due to the rise in energy prices rather than the EU ETS. Decomposing the allowance surplus allows assessing overallocation and thus overallocation profits. We estimate that the cement industry reaped 3.5 billion euros of overallocation profits during phases I and II, mainly because of the slowdown in production, while allowance caps were unchanged. This figure re-

[5] As long as they are allocated independently of current output (which is the case except partially for 2012, see Chapter 5), the operator of an installation may reduce emissions in order to sell allowances even though he has received more allowances than its emissions (Montgomery, 1972). Free allocation would be inefficient at preventing leakage in the short term and would only provide a disincentive to plant relocation (Wooders et al., 2009).

lated to production represents a very significant amount of the margins observed in the sector (Boyer and Ponssard 2013). Presented at first as a threat to competitiveness, the EU ETS has paradoxically boosted European cement industry profitability.

European authorities modified the allocation rules with the intention to reduce excess free allocation in low-activity plants. After 2012, whenever the activity level of an installation falls below some threshold (50%, 25%, 10%) relative to its historical activity level, the allocation would be reduced accordingly (50%, 25%, 0%). For installations operating below the threshold, the financial gain from "artificially" increasing output to reach the threshold may outweigh the costs, particularly in carbon intensive sectors. We show in Chapter 5, exploiting the constituted EUTL cement installation-level database, that cement companies strategically reacted to this activity level thresholds (ALTs) rule in order to maintain a high amount of free allocation. The quantification of distortions due to the thresholds necessitates the elaboration of a counterfactual scenario for 2012, which is developed by using a panel data model combining historical data at the country and plant level. We estimate that in 2012, ALTs induced 6.4Mt of excess clinker production (5% of total EU output), which corresponds to 5.8Mt of excess CO2 emissions (over 5% of total sector emissions). The distortion effects are magnified in crisis-hit countries with low demand, especially in Spain and Greece. As intended, ALTs do reduce overallocation (by 6.4 million allowances) relative to a scenario without ALTs, but this gain is small compared to an output based allocation method, which would further reduce overallocation by 40 million allowances (29% of total cement sector free allocation). We then show, revisiting preliminary evidence from Neuhoff et al. (2014), that in order to avoid distubing local markets while increasing production, cement companies (i) shifted production among nearby plants, (ii) exported clinker or cement to other markets (iii) increased the clinker-to-cement ratio. As the decrease of the latter was the main levy of emissions reduction in the sector (see Chapter 4), the operational distortions due to the introduction of ALTs are particularly detrimental. These considerations suggest that the ALTs rule may need to be reconsidered for sectors such as cement for which carbon costs represent a significant share of production costs. A relatively easy-to-implement solution for the short term would be to implement full output-based allocation.

The main argument of this thesis is that competitiveness, which was called a "dangerous obsession" for macroeconomic policy by Krugman (1994), may be so for climate policy as well. The pursuit of carbon leakage mitigation has contributed to make the EU ETS an administrative nightmare which fails to give a clear signal towards low carbon transition. In the conclusion, we will discuss possible reforms and propose directions for future research.

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1

Climate policy and the "carbon haven" effect

The Kyoto Protocol has been an attempt to set a global climate architecture aimed at abating carbon emissions on a global scale. The commitment period of the protocol ended in 2012 with mixed results. While abating technologies have improved, in particular renewable energies, the world's CO_2 emissions reached a record in 2011 with 31.6 Gt [1] , an increase of 50% compared to 1990 emissions, and are likely to keep increasing in the next decade. Despite the growing emergency of serious climate change impacts, international negotiations are blocked because of strong free-riding incentives (Carraro and Siniscalco, 1993), lobbying from energy intensive sectors and equity concerns about the North-South burden sharing. Climate policies will remain sub-global in the years to come, and unilateral or regional policies, including regulations, subsidies, carbon taxes and carbon

[1] `https://www.iea.org/newsroomandevents/pressreleases/2013/june/four-energy-policies-can-keep-the-2c-climate-goal-alive.html`

markets, have emerged as some industrialized countries decided unilaterally to reduce their emissions. The top-down global Kyoto approach is shifting towards a bottom-up architecture with different CO_2 prices (Rayner, 2010; Weischer et al., 2012).

In a world with uneven climate policies, the carbon price differentials across regions modify production costs and may shift the production of energy-intensive goods from carbon-constrained countries to "carbon havens", or countries with laxer climate policy. Since a decrease in emissions in one part of the world leads to an increase in emissions in the rest of the world, this phenomenon is referred to as carbon leakage. The Pollution Haven effect, that is, the migration of dirty industries to countries with less stringent regulations, is one of the most contentious debates in international economics (Taylor, 2005). A major difference exists between local pollutants, which constitute the overwhelming part of studies in the pollution haven literature, and CO_2. CO_2 is a global stock pollutant: the geographic location of emissions does not matter (Siikamäki et al., 2012b). A production shift would then reduce the environmental benefits of the policy while potentially damaging the economy.

In the context of growing globalisation, environmental policies can also have a strategic role. The fierce competition to attract foreign direct investment or the threat of industrial relocation could lead to a "regulatory chill" or even a "race-to-the bottom", depending on the willingness of countries to downgrade environmental standards. Indeed, the fear of carbon leakage and loss of competitiveness in energy-intensive industries are the main arguments against ambitious climate policies in industrialized countries. Modest mitigation targets have gone hand in hand with policy packages intended to protect sectors at risk of carbon leakage (mainly cement, iron and steel, aluminium and oil refineries). In the European Union Emission Trading System (EU ETS), the biggest carbon pricing experiment so far, tradable allowances are distributed free of charge for these sectors . In the US, the Waxman-Markey proposal, which was adopted by the House of Representatives in 2009 but not by the Senate, would have introduced a nationwide carbon market with measures to face these issues: allowances distributed freely on the basis of current output (output-based allocation) and border carbon adjustment (BCA). The latter, aimed at "levelling the carbon playing field", is widely discussed among politicians, business leaders and academics. However, it is often considered as protectionism disguised as green policy (Evenett and Whalley, 2009) among developing countries, and its World Trade Organisation compatibility remains contentious. The political outcome of its implementation is highly uncertain. BCA may increase the incentives of third countries to join the abating coalition but may also create international friction and lead to tit-for-tat trade retaliations (Bordoff, 2009; of Foreign Trade, 2010). The recent setbacks of the inclusion of aviation in the EU ETS are a reminder that any attempt to regulate emissions outside a country's jurisdiction is extremely problematic: foreign airlines and governments complained aboutthis inclusion, which pushed the EU to delay the inclusion of international flights by one year. Whether this inclusion will take place at the end of the delay period is still unclear.

This chapter provides a literature review on competitiveness and carbon leakage issues from an economic, political and legal perspective. First, section 1.1 gives the definition of the main terms involved. Section 1.2 provides an evaluation of the carbon leakage risk, distinguishing ex ante Computable General Equilibrium (CGE) modelling from ex post econometric studies. Section 1.3 examines the policies aimed at reducing carbon leakage and competitiveness losses with an emphasis on Border Carbon Adjustment. Since the consistency of BCA with WTO is a decisive matter, it is discussed in further detail in section 1.4. Section 1.5 concludes.

1.1 Definitions

1.1.1 Carbon leakage

While competitiveness concerns and carbon leakage are often associated, they are two distinct phenomena. Carbon leakage is the increase of emissions in the rest of the world when a region implements a climate policy, compared to a situation where no policy is implemented (Quirion, 2010). It can be measured by the leakage rate or leakage-to-reduction ratio, which is the rise in emissions in the rest of the world divided by the abated emissions in the region that has adopted a climate policy. A 50% leakage-to-reduction ratio means that half of the mitigation effort is undermined by the increase of emissions in the rest of the world, and not the misguided interpretation that 50% of emissions have "leaked" in the rest of the world. If this ratio is under 100%, emissions have decreased on a global scale, so the policy is environmentally beneficial. A ratio above 100% is theoretically possible, because the carbon intensity of CO_2-intensive products can be higher in the rest of the world, but has only been found in one outlier model (Babiker, 2005). Estimates of leakage rates are typically in a range of 5%-20% depending on many factors (see below). Carbon leakage occurs through two main channels: the competitiveness channel and the international fossil fuel price channel (Dröge, 2009). The root of the competitiveness channel is that the cost of compliance gives a comparative disadvantage for regulated firms vis-à-vis their competitors. This change of relative prices can lead to a change of the trade balance (less exports and more imports). In the short term, this would correspond to a change of the utilisation rate of existing capacities (operational leakage), while in the long term, it would correspond to a change in production capacities (investment leakage). These changes induce a shift of production, and then of emissions, from the regulated part of the world to the unregulated part of the world. Besides, abating countries almost necessarily have to cut their fossil fuel consumption, which drives down the international prices of carbon-intensive fossil fuels: coal, oil and, perhaps even more, non-conventional fossil fuels (Persson et al., 2007). This decrease in prices reduces the net cost of climate policies in fuel-importing abating countries since a part of abatement is borne by fossil fuel exporters who lose a part of their rents. However it leads to a rise of their consumption in countries with less stringent policies. Because of international energy markets, the shrink in consumption in one region involves an increase in consumption in the rest of the world, causing carbon leakage through the international fossil fuel price channel. Yet two caveats are in order. First, CO_2 capture and storage (CCS) does not reduce fuel consumption. Quirion et al. (2011) show that for this reason, CCS brings down carbon leakage compared to a climate policy providing the same abatement without CCS. Second, the world oil market is dominated by OPEC, and alternative assumptions about OPEC's behaviour lead to opposite results regarding leakage through the oil market, which can even become negative (Bohringer et al., 2014). The same reasoning applied to the whole world but with two temporal periods is known as the Green Paradox (Sinn, 2008; Eisenack et al., 2012) which could be considered inter-temporal leakage: a rising CO_2 price would be seen as a future resource expropriation by fossil fuel owners who would then increase resource extraction. Yet, although the mechanism of the Green Paradox is well understood, its quantitative importance decreases when realistic features are included in the models (Gerlagh, 2010). Despite the overwhelming importance of the competitiveness channel in the climate policy debate, in virtually all models including the two channels, the international fossil fuel price channel predominates (Gerlagh and Kuik, 2007; Fischer and Fox, 2012; Boeters and Bollen, 2012; Weitzel et al., 2012).

1.1.2 Competitiveness

The term "competitiveness" has been used in numerous studies, reports and articles and underlies economic policies. However, this concept is difficult to define and susceptible to ambiguities. At a firm or sectoral level, competitiveness can refer to "ability to sell" or "ability to earn". Competiveness as "ability to sell" is the capacity to increase market share, and can be measured through indicators involving exports, imports and domestic sales (Alexeeva-Talebi et al., 2007). Competitiveness as "ability to earn" is the capacity to increase margins of profitability, and can be measured with indicators involving some measures of profit or stock values. Distinguishing these two notions is useful since the same climate policy can have different impacts on both. For instance, distributing free emission allowances based on historic data only, as is the case in the US SO2 ETS (Schmalensee and Stavins, 2013), increases the ability to earn but not the ability to sell, since an operator can close a plant and continue to receive the same amount of allowances. Hence, only competitiveness as ability to sell may generate leakage.

The notion of competitiveness at the national level is controversial, and is considered meaningless by some economists, such as (Krugman, 1994). The main indicator is the balance of trade, that is, the difference between the monetary value of exports and imports, but an increase in the balance of trade may result from many factors, some of which are completely unrelated to the competitiveness of domestic firms, like a contraction in domestic demand. Whether climate policies have to protect competitiveness at a national level or at a sectoral level is a legitimate question. EU ETS sectors contribute 40% of EU emissions, but less than 5% of its Gross Domestic Product (GDP) and an even smaller share of its jobs (Ellerman et al., 2010). The sectors at risk of carbon leakage (see below) account for slightly more than 1% of GDP in the UK (Hourcade et al., 2007) and 2% in Germany (Graichen et al., 2008). However, they account for a much higher share of Greenhouse Gas (GHG) emissions so protecting their competitiveness in order to limit leakage cannot be discarded prima facie.

1.1.3 Sectors at risk

All sectors do not face the same risk of carbon leakage. The risk is higher if the carbon cost is high and the international competition is fierce. Hence, in the attempt to classify sectors exposed to carbon leakage, two indicators are generally used, one measuring the carbon cost and the other the trade intensity. For the EU ETS, the carbon cost is measured by the value at stake, defined as the carbon costs relative to the gross value added of a given industrial sector. The trade intensity is measured by the ratio in values between imports plus exports and the EU total market size. A sector is considered at risk if one or both of these indicators is above a certain threshold (see Figure 1.1). Table 1 shows the different indicators and thresholds to identify sectors at risk in the EU, the US and Australia. The most vulnerable sectors, usually gathered around the common denomination of Energy Intensive Trade Exposed (EITE) sectors, include iron and steel, cement, refineries and aluminium. The EITE sectors are well-organized and constitute a strong lobby that has managed so far to influence climate policies. Indeed, all climate policies have provided more favourable rules for these sectors compared to others. In addition, these "specific rules" are generally more favourable in the final amendments than in first drafts (CEO, 2010). The classification of sectors in itself (which sectors are at risk and which are not), because of its economic impacts, is subject to political and academic controversy and face strong industrial lobbying (Clò, 2010; Martin et al., 2014).

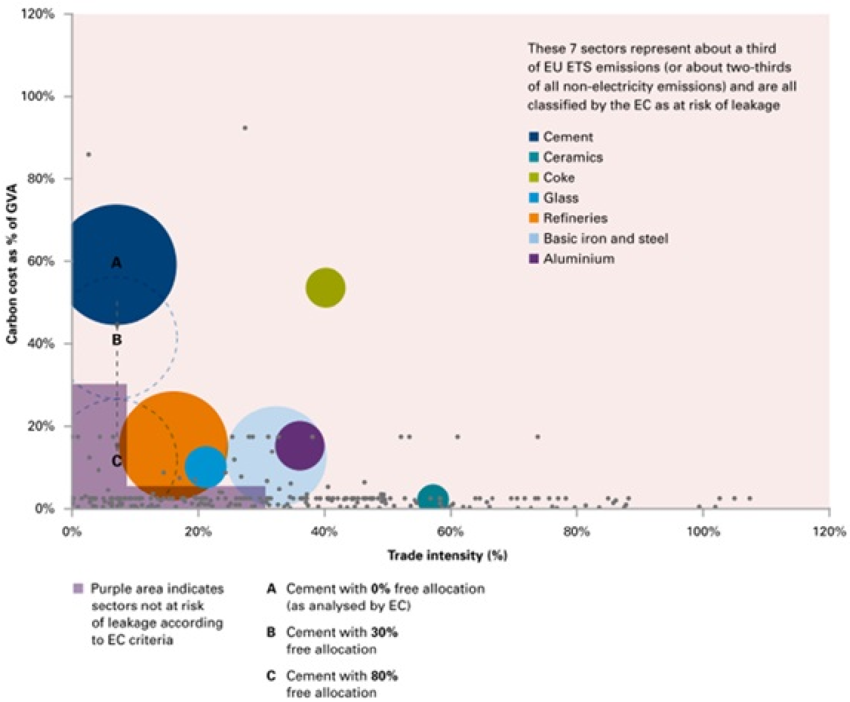

Figure 1.1: Sectors classified "at risk of carbon leakage" in Europe (source: Grubb and Counsell (2010)). The size of the circles is proportional to the sector emissions.

1.1.4 Positive impacts of climate policies on competitiveness and abatement in foreign countries

Though the political debate has focused on the negative impacts of climate policies, some authors argue that at least in some sectors or firms, stringent environmental regulations can force firms to be more efficient in their processes, and then more competitive. This is referred to as the Porter hypothesis (Porter and Linde, 1995), which is highly controversial but has been corroborated in Europe by a recent econometric study (Costantini and Mazzanti, 2012). Further, it is possible to highlight two mechanisms symmetrical of carbon leakage and competitiveness losses: climate spillovers and first mover advantage.

Environmental regulations foster innovation and generate technological progress in GHG savings technologies (Newell et al., 1999; Jaffe et al., 2002; Dechezleprêtre et al., 2008). Diffusion of these technologies reduces emissions in non-abating countries and then creates negative leakage, or positive climate spillover (Gerlagh and Kuik, 2007; Maria and Werf, 2008; Golombek and Hoel, 2004; Bosetti et al., 2008). There is empirical evidence of climate spillovers, especially in energy-saving technologies (Popp, 2002), but also in renewables. Feed-in tariffs in Denmark, Germany and Spain generated a massive induced technical change in wind and solar technologies (Peters et al., 2012) and are thus in part responsible for the spectacular development of windpower capacities in China, which became the world leader in terms of windpower installed capacities, shifting from 2.6 GW in 2006 to 75 GW in 2012 (Roney, 2013). Another, yet even more difficult to quantify source of negative leakage is the international diffusion of climate policies: implementing any new policy involves some risks, and observing climate policies in other countries allows reducing these risks and possibly avoiding some mistakes. Just as the EU has closely observed the US SO2 cap-and-trade to set up the EU ETS, subsequent ETS developments have benefited from the EU ETS experience. The same stands for other climate policies such as renewable subsidies (especially feed-in-tariffs pioneered by Denmark and then Germany) and energy efficiency regulations.

Finally, Baylis et al. (2014) have recently identified a new mechanism generating negative leakage, which they label the Abatement Resource Effect (ARE).

The intuition is that when a climate policy reduces emissions in one part of the economy, it may draw factors of production away from other, carbon-intensive activities. The authors show that if this effect is strong enough, an economy may exhibit negative net leakage in response to the policy change. While the possibility of negative leakage through this mechanism is not disputed, (Carbone, 2013) as well as (Winchester and Rausch, 2013) have recently assessed the ARE in more complex models and conclude that the negative leakage due to the ARE is more than offset by positive leakage mechanisms.

Technological knowhow in climate-related technologies gained by domestic firms could be used to capture market share in emerging markets (first-mover advantage). If other countries join the abating coalition, these firms have a comparative advantage vis-à-vis their competitors. This ability to gain market share by being the first to develop a technology is the first mover advantage. Emerging in models (Pollitt et al., 2015), it could be considered a long-term competitiveness factor. The clearest case concerns the EU wind industry, which is the dominant supplier in all word markets except China, due to the already mentioned feed-in-tariffs implemented in the 1990s. However, while Germany benefited from a first-mover advantage the Photovoltaic (PV) industry until 2011, the German PV industry has since been largely surpassed by China, showing how fragile a dominant position can be in industries featuring fast technical progress (Kazmerski, 2011).

1.2 Evaluation of carbon leakage

1.2.1 Ex ante studies

Climate change mitigation policies are diverse and include various forms of regulations, subsidies, carbon taxes and emission trading systems (ETS). Yet carbon leakage has mostly been assessed for ETS and carbon taxes. There is extensive literature assessing ex ante carbon leakage from hypothetical carbon taxes or ETS that can be traced back to Felder and Rutherford (1993). The majority of these studies rely on Computable General Equilibrium (CGE) models (Böhringer et al., 2012a; Mattoo et al., 2009; Fischer and Fox, 2012; Dissou and Eyland, 2011; Lanzi et al., 2012; Balistreri and Rutherford, 2012; Peterson and Schleich, 2007), but some use partial equilibrium models (Gielen and Moriguchi, 2002; Mathiesen and Maestad, 2004; Monjon and Quirion, 2011; Demailly and Quirion, 2006, 2008). CGE models, which simulate the behaviour of entire economies, are pertinent to study the effect of policies on trade in different sectors (Kehoe et al., 2005) but they generally rely on more aggregated data (almost exclusively the Global Trade Analysis Project database) that may hide impacts on more specific sectors (Siikamäki et al., 2012a; Alexeeva-Talebi et al., 2012). Moreover, most CGE models feature a zero-profit condition so cannot assess competitiveness as ability to earn. An exception is Goulder et al. (2010) whose model features capital adjustment costs, which implies that capital is imperfectly mobile across sectors and allows the model to capture the different impacts of policy interventions on the profits of various industries. Assessing a hypothetical federal ETS in the US, the authors conclude that freely allocating fewer than 15% of the emissions allowances generally suffices to prevent profit losses in the most vulnerable industries. Freely allocating all of the allowances substantially over-compensates these industries.

These models provide a wide range of estimations for leakage and competitiveness losses (as ability to sell). First, results depend on scenario hypotheses: the bigger the abating coalition, the smaller the leakage rate while the more ambitious the target, the higher the leakage. Linking carbon markets within the abating coalition (Lanzi et al., 2012), authorizing offset credits (Böhringer et al., 2012a) or extending carbon pricing to all GHG (Ghosh et al., 2012) increases economic efficiency and then reduces leakage. Second, the models are very sensitive to two sets of parameters: fossil fuel supply elasticities (for the international fossil fuel price channel) and Armington elasticities (for the competitiveness channel) (Monjon and Quirion, 2011; Alexeeva-Talebi et al., 2012; Balistreri and Rutherford, 2012). The former indicate to what extent a decrease in fossil fuel demand reduces the fuel price, while the latter represent the substitutability between domestic and foreign products.

A recent comparative study of 12 different models gave the most robust results so far (Böhringer et al., 2012a). The estimate of leakage is 5-19% (mean 12%) when Annex I countries (except Russia) abate 20% of their emissions through carbon pricing without taking any measure to protect EITE sectors. The loss of output in these sectors is 0.5%-5% (mean 3%) in the coalition and an output gain of 1%-6.5% (mean 3%) is observed in the rest of the world. A detailed review of leakage estimates is provided in chapter 2. These aggregate results hide differences among sectors, but even at sectoral levels, leakage estimates contrast sharply with alarmist predictions made by industry-financed studies. For example, according to a (BCG, 2008) study funded by the European cement industry, under carbon pricing at 25 euro /tCO_2 without climate policy outside the EU ETS or measures against leakage, importers would supply 80% of the European cement market. A peer-reviewed study that analyses a very similar scenario (except that the CO_2 price is at 20 euros per ton) concludes that importers would only supply 8%, versus 3% absent climate policy (Demailly and Quirion, 2006). These contrasted results can be explained by different assumptions about available production capacities abroad and the nature of competition assumed in the cement market.

1.2.2 Ex post studies

The first studies assessing empirically the impacts of environmental regulations on trade dealt with local pollution issues (Kalt, 1988; Tobey, 1990; Grossman and Krueger, 1993; Jaffe et al., 1995). They showed little evidence to support the "pollution haven" effect: their estimates of the impact of environmental regulations on trade flows were either small or insignificant. However, recent studies have shown some evidence of the pollution haven effect in small proportions (Dean et al., 2005; Levinson and Taylor, 2008). Paradoxically, dirty industries seem less vulnerable, because of capital intensity and transport costs (Ederington et al., 2003). The empirical validity of the pollution haven effect continues to be one of the most contentious issues in the debate regarding international trade and environment (Kellenberg, 2009). Nevertheless a massive environmental relocation has never been observed.

Environmental tax reforms (ETR, i.e. carbon taxes whose revenues are used to cut other taxes, mostly on labour income) established in some European countries offer another natural experiment to empirically treat these questions. Kee et al. (2010) analyse the evolution of imports and exports in energy-intensive industries, comparing countries which did and did not implement a carbon tax. The authors find a statistically significant negative impact on exports of a carbon tax only in the cement sector while, strangely enough, they find a positive impact on exports in the paper as well as iron and steel sectors. No statistically significant impact was found on imports for any sector. (Miltner and Salmons, 2009) found that, out of 56 cases (seven countries and eight sectors studied), the impact of ETR on competitiveness was insignificant in 80% of the cases, positive in 4% and negative for only 16%. However, EITE sectors benefited from exemptions and lower taxation rates, which may explain why more negative impacts were not observed. If ETR didn't prove harmful for these industries, they had a positive impact on economic wealth, giving empirical arguments for the double dividend theory (Barker et al., 2009), e.g. a taxation shift from labour to pollution may stimulate economic growth as well as reducing pollution (Goulder, 2002; Bento and Jacobsen, 2007).

Aichele and Felbermayr (2012) econometrically assessed the impact of having an emission target under the Kyoto Protocol (i.e. being a developed country and having ratified the Protocol) on CO_2 emissions, the CO_2 footprint[2] and CO_2 net imports, using a differences-in-differences approach on a panel of 40 countries. To account for a potential endogeneity bias (the fact that countries with an expected low or negative growth in emissions may be more likely to have ratified the Protocol) they use the International Criminal Court participation as an instrumental variable for Kyoto ratification. They concluded that countries with a Kyoto target reduced domestic emissions by about 7% between 1997-2000 and 2004-2007 compared to the countries without a target, but that their CO_2 footprint did not changed (CO_2 net imports increased by about 14%). These results imply that domestic reductions have been fully offset by carbon leakage. However two caveats are in order. First, China became a member of the WTO in 2002, just when most developed countries ratified the Protocol. Since most CO_2 net imports are due to trade with China (Sato, 2013), the rise in net imports may well be due to China WTO membership rather than to Kyoto. Second, apart from those covered by the EU ETS, countries with a Kyoto target haven't adopted significant policies to reduce emissions in manufacturing industry. Hence, if Kyoto had caused leakage (through the competitiveness channel), it should show up on the CO_2 net imports

[2]The CO_2 footprint equals domestic emissions plus CO_2 net imports, i.e. domestic emissions plus emissions caused by the production of imported products, minus emissions caused by the production of exported products.

of countries covered by the EU ETS rather than of countries covered by a Kyoto target; yet the authors report that EU membership does not increase CO_2 imports, when they include both EU membership and the existence of a Kyoto target in the regression. This conclusion invites to look more directly at the impact of the EU ETS.

The studies focusing on the EU ETS, the largest carbon pricing experiment so far, have not revealed any evidence of carbon leakage and loss of competitiveness in sectors considered at risk of carbon leakage, such as cement, aluminium, and iron and steel (Reinaud, 2008; Ellerman et al., 2010; Sartor, 2013; Quirion, 2011). More studies will undoubtedly be conducted in the following years, for the EU ETS and the other carbon markets that have emerged, as more hindsight will be provided. So far, the empirical resultsare in sharp contrast to the "exodus of EU industry" claimed by the European Alliance of Energy Intensive Industries (Oxfam, 2010).

1.2.3 Synthesis

Ex ante modelling studies vary in their results because of policy scenarios (size of the coalition, abatement targets) and some crucial model parameters (Armington elasticities for the competitiveness channel, and oil supply elasticities for the international fossil fuel channel). A meta-analysis of recent studies which details the role of these factors is provided in Chapter 2. In the absence of BCA, most of these studies suggest leakage rates in the range of 5-20%. Conversely, ex post econometric studies have not revealed empirical evidence of these issues. Why such a difference?

First, effects of carbon taxation are always in practice compensated by "policy packages". Because of carbon leakage and competitiveness concerns, sectors at risk in the EU ETS received allocations free of charge while in every case of CO_2 tax, they benefited from lower tax rates or exemptions. In addition, aluminium producers and other electricity-intensive industries, protected by long term electricity contracts, have not always suffered the pass-through of carbon costs to consumer by electricity companies (Sijm et al., 2006). Moreover, in the case of the EU ETS, the CO_2 price has been below 14 euros for the majority of the time since the launch of the system, arguably too low a value to entail noticeable impacts. Further, empirical studies have focused so far on operational leakage and not investment leakage (change in production capacities), which could be studied through the analysis of foreign direct investments. Over time, new carbon markets are launched and time series get longer, giving more room for empirical research. However, assessing the "true" impact of asymmetric carbon pricing will always be hampered by the compensation measures aimed at reducing competitiveness losses. Another reason for the gap between ex ante predictions and ex post analysis could be that models generally do not (or only vaguely) take into account positive aspects of climate policies, such as climate spillover and first mover advantage. More research understandings of the positive aspects of climate policies would be useful when exploring the climate and competitiveness linkages. Other possible areas of improvement is further contribution to the empirical literature, which remains thin, and progress in international trade theories.

1.3 Policies to address leakage and competitiveness concerns

The elaboration of policy tools designed to "level the carbon playing field" has led to an extensive body of literature. One can classify these measures in three broad categories: a global approach, levelling down the cost of carbon and border adjustments (Grubb and Counsell, 2010; Dröge, 2009). Each of these categories has many variants and a combination of different tools could also be considered. The next sections discuss their specific features, pros and cons. None of these instruments seems to be a "magic bullet" to address both economic efficiency, equity and practical feasibility concerns (Böhringer et al., 2012a). Some argue that policies to address this problem should be sector-specific (Grubb and Counsell, 2010; Dröge, 2009), but so far tools that have actually been implemented or considered to address competitiveness and leakage concerns only distinguished sectors "at risk" from the others: see Figure for Europe (EU ETS phase II and III), the US (Waxman-Markey amendment), Australia (Clean Energy Legislative Package), the California ETS and the New Zealand ETS.

	EU ETS Phase I	EU ETS Phase II	U.S. Waxman-Markey Act (H.R 2454)	Australian CELP	California ETS	New Zeal ETS
Eligibility indicators	None (all sectors receive free allocation)	Two indicators: Trade Intensity (TI) ((Imports + Exports)/(Turnover + Imports)) and Value-at-Stake (VaS) (CO_2cost/Gross Value Added)	Two indicators: Trade Intensity (TI) and Carbon Cost/Value of Shipment (CC)	Three indicators: Trade Intensity (TI), Emissions Intensity in Revenues (EIR) and Emissions Intensity in value-added (EIVA)	Two indicators: Emissions Intensity (EI) (tCO_2e/$millions value-added) and Trade Exposure (TE), (equivalent of Trade Intensity)	One indic Intensi (manu are co trade- there i trade)
Eligibility criteria	None (all sectors receive free allocation)	(i)VaS > 5% and TI > 10% (ii) VaS > 30% (iii) TI > 30%	(i)TI > 15% and CC > 5% (ii) CC > 20%	First tier: (i) TI > 10% and EIR > 2000tCO_2/ AUDm or (ii) EIVA > 6000tCO_2/ AUDm Second tier: (iii) TI > 10% and 1000 < EIR < 2000tCO_2/ AUDm or 3000 < EIVA < 6000tCO_2/ AUDm	3 categories of leakage risk: high, middle and low. High: (i) EI > 5000 or (ii) EI > 1000 and TE > 19% Medium (iii) 5000 > EI > 1000 and TE < 19% or (iv) EI > 100 and TE > 10%	Highly En EI > 1 million Emissi EI > 8 NZ$
Initial free allocation level	100%	100%	0% (but output-based rebates)	First tier: 94.5% Second tier: 66%	100% for High leakage risk from 2013 to 2020—for Medium leakage risk 100% for 2013–2014, 75% for 2015–2017 and 50% for 2018–2020	90% for for Mo
Allocation method	Mainly historic emissions (National Allocation Plans)	10% best ETS average emissions of installations of a given sector.	Average industrial emissions	Average industrial emissions Decline 1.3% a year ('carbon productivity contribution')	'Product-based' benchmark. The cap is declining by roughly 0.9% per year for emission intensive manufacturing sectors	'Intensity Bench declini
Output-based allocation feature	No	No	Output-based rebates for sectors at risk	Corrections in the global cap to take into account production	Yes	Yes
Subsidies	No	No	No	AUD 300 million Steel Transformation Plan AUD 1.3 billion Coal Sector Jobs Package	No	No
Border Carbon Adjustment	No	No	Yes, after 2020, with exceptions	No	Emissions from imported electricity are covered	No

Figure 1.2: Policy packages for sectors at risk of carbon leakage (source: Hood (2010); Spencer et al. (2012)).

1.3.1 Global approach

The first-best solution would be the existence of a uniform carbon price allowed by international climate agreements and flexibility mechanisms. However, because of the negative perspective of international climate negotiations, this option seems highly unlikely until at least 2020.[3] A pragmatic alternative would then be to embrace cooperative sectoral approaches (Houser, 2008; Zhang, 2012; Hamdi-Cherif et al., 2011) but much confusion remains regarding what they should be. Developed countries favour the form of industry targets and timetables, and diffusion of performance standards, thus addressing leakage and competitiveness concerns. Conversely, developing countries such as India are suspicious of the imposition of binding targets through sectoral approaches and interpret sectoral agreements as a catalyst for technology transfer (Meckling and Chung, 2009).

1.3.2 Levelling down the cost of carbon

Levelling down can be achieved through investment subsidies, sectoral exemptions or free allocation of permits, so as to decrease or even suppress the carbon cost for targeted sectors. All are equivalent to subsidies, and are then subject to the agreement on Subsidies and Countervailing Measures (SCM) of the World Trade Organization. Exempting the most vulnerable sectors was implemented in Norway (Bruvoll and Larsen, 2004) and Sweden (Johansson, 2006) when their carbon tax were introduced. It solves the competitiveness and leakage concerns but at a substantial economic cost (Rivers, 2010; Böhringer et al., 2012b): since emissions in these sectors will not be reduced, to reach a given aggregate target, more abatement must take place in the others, including less cost-effective options.

Instead of auctioning, three main options for allocating free allowances have been considered: historic, output-based and capacity-based allocation (used in the EU ETS). These free allocation methods induce side effects: in order to prevent competitiveness issues, other distributional and cost-effectiveness issues are created. In case of historic and capacity-based allocation the ability to pass-through

[3]The goal of international negotiations is to sign international agreements before 2015 that would be implemented after 2020.

carbon costs creates windfall profits for the operators of covered installations (Sijm et al., 2006; Sandbag, 2012). Nevertheless, simulations indicate that output-based allocations seem more efficient to counteract leakage and protect industrial competitiveness while assuring political acceptability (Quirion, 2009; Rivers, 2010).

1.3.3 Border adjustments

Border Carbon Adjustments (BCA) consist of reducing the carbon price differentials of goods traded between countries, inspired by measures in place for Value Added Tax. Based on theoretical grounds to improve the cost-efficiency of sub-global climate policies (Markusen, 1975; Hoel, 1996), BCA were also considered a way to "punish" the US for free-riding the Kyoto Protocol (Hontelez, 2007). Later, the US incorporated BCA in the Waxman-Markey amendment, aiming mainly at Chinese products (van Asselt and Biermann, 2007). However the fierce criticism of China and India led President Obama to dissociate the US administration from this proposal (declaring "We have to be very careful about sending any protectionist signals", (Broder, 2009)). Among the advocates of BCA, one can cite Paul Krugman (Krugman, 2012), who argues that BCA are "a matter of levelling the playing field, not protectionism".

Many technical points are to be considered for the implementation of BCA (Cosbey et al., 2012; Monjon and Quirion, 2011), which are not inconsequential technical details, but would determine the viability of this option under international laws:

- Covered sectors. There is a general consensus that only sectors at risk should be covered by the scheme; however, the classification of sectors at risk may be controversial (for example for the third phase of EU ETS, see Clò (2010); Martin et al. (2014)).

- Covered countries. Country exceptions may occur, for example, for Least Developed Countries for equity purposes or, as in the Waxman-Markey bill, for countries that have taken "comparable action" on climate policies. However climate policies are so various, being a mix of carbon pricing, regulation and subsidies, that comparing different climate policies is not easy. One can distinguish two principles: "comparability in effectiveness" as in the WTO Shrimp-Turtle dispute or "comparability of efforts" as in the Common but Differentiated Responsibilities principle.

- Inclusion of indirect emissions. Taking into account the indirect emissions from electricity consumption is relevant for industries with high electricity costs, such as aluminium, but highly complicates the calculation of adjustment factors. The energy mix differs among countries, and calculation of emissions from electricity consumption is contentious, because of differences between marginal and average specific emissions.

- Inclusion of export rebates. They are useful to level the playing field also in third countries markets, but their WTO compatibility is not guaranteed.

- Carbon content. One can consider four options: exporter's average emissions, home country's average emissions, self-declaration or best available technology (BAT) based on benchmarks. A reliable knowledge of the carbon content of every foreign product seems out of range because of information asymmetry and administrative costs. To avoid a WTO challenge because of discrimination, these estimations should be rather conservative, which favours BAT benchmarking, or a choice between home country's average emissions and self-declaration (Ismer and Neuhoff, 2007).

- Legal form of the adjustment. The adjustment could take the form of a tax or of an obligation to surrender allowances. The origin of these allowance is to be determined (home region or under UNFCCC, with the possibility or not to come from offset credits).

- Use of revenues. The share of revenues between the importing country, the exporting country and an international body to be designated is crucial and may be the biggest levy of political acceptability. Many have argued that these revenues could be used to finance clean technology transfer or adaptation through a Green Climate Fund (Godard, 2009; Grubb, 2011; Springmann, 2013).

- Timing. A period of good faith could be offered to third countries before the implementation of such measures. Clear conditions for phasing out must also be decided.

Among all these features, some are incorporated as scenario alternatives in models, such as the covered sectors (Ghosh et al., 2012; Peterson and Schleich, 2007; Mattoo et al., 2009; Winchester et al., 2011), the inclusion of indirect emissions (Böhringer et al., 2012b; Monjon and Quirion, 2011), the inclusion of export rebates (Lanzi et al., 2012; Fischer and Fox, 2012; Bednar-Friedl et al., 2012), the carbon content (McKibbin et al., 2008; Kuik and Hofkes, 2010) and the use of revenues (Boeters and Bollen, 2012; Böhringer et al., 2014; Rivers, 2010). However, both technical difficulties and administrative costs (as input-output matrices for carbon content are "available" in models) and legal challenges (as they go beyond energy-economy modelling) are under-evaluated in these models.

Border adjustments are effective to reduce leakage through the competitiveness channel (but obviously not leakage through the international fossil fuel price channel): in model simulations, the leakage rate decreases by about 10 percentage points on average (Böhringer et al., 2012a). They are also very effective to protect competitiveness but they shift a part of the mitigation burden to developing countries (Bao et al., 2013). With a CGE model, Mattoo et al. (2009) find that strong BCA imposed by US would depress India and China manufacturing exports between 16% and 21%. However, it must be remembered that China will in all likelihood consume domestically more than 98% of its steel production[4] and 99% of its cement production:[5] the effects of BCA on Chinese production would then be very small. BCA might conflict with the Principle of Common but Differentiated Responsibilities of the UNFCCC (Dröge, 2011).

Its effect on international negotiations is unclear: they could be used as a "strategic stick" to force other countries to join the abating coalition (Lessmann et al.,

[4]In 2007 (and, respectively, 2011), China produced 489 Mt (resp. 684 Mt) of steel and exported 50 Mt (resp. 13 Mt). Therefore China consumed 90% of its production in 2007 and 98% in 2011 (source `http://www.issb.co.uk/asia.html`). Steel production is expected to boom whereas exportations are expected to stay in the same level.

[5]China produced 2 Gt of cement in 2011 and exported 15,6 Mt in 2009 (we suppose the exports in 2011 have the same magnitude), meaning that China consumed 99% of its production. source `http://www.globalcement.com/news/itemlist/tag/Chinaandhttp://www.articlesbase.com/business-articles/chinese-cement-industry-realized-the-sales-of-cny-50072-billion-in-2009-1937146.html`

2009), but they could also trigger a trade war because of "green protectionism" suspicions (of Foreign Trade, 2010). For example, China strongly opposes BCA and claims that energy-intensive exports are already taxed (Voituriez and Wang, 2011). Climate coalition countries have an incentive to deviate from the optimal carbon tariff rate to change their terms of trade (Weitzel et al., 2012), and even with good-quality data, there is room for judgement discretion in carbon content estimation and hence disguised protectionism (Holmes et al., 2011).

Some argue that the "carrot" of technology transfer would be more effective than the "stick" of BCA (Weber and Peters, 2009). Further, the benefits of internal improvements of emission trading systems within the abating coalition like linking markets and extending sectoral coverage could outweigh those of BCA (Springmann, 2012; Lanzi et al., 2012). Finally, the most controversial aspect of this measure is its compatibility with the WTO, discussed in the next section.

1.4 Border Carbon Adjustments and the World Trade Organization

The General Agreement on Tariffs and Trade (GATT) was established in a world without climate change on the international agenda, so its rules were not drafted to address climate policies, making the interpretation of legal texts particularly difficult. Past WTO cases, such as the Superfund, Tuna-Dolphin and Shrimp-Turtle reveal some information, but many features of BCA are unprecedented and WTO panels are not bound by previous decisions (no rule of stare decisis) (Zhang and Assunção, 2004). Hence, assessing the WTO consistency of BCA according to its specific features divides legal experts and has led to extensive literature on the subject (Biermann and Brohm, 2004; Goh, 2004; Frankell, 2005; Cendra, 2006; van Asselt and Biermann, 2007; Ismer and Neuhoff, 2007; Pauwelyn, 2007; Green and Epps, 2008; Sindico, 2008; Quick, 2008; Bordoff, 2009; Low et al., 2011; Zhang, 2012). If there is a consensus among legal experts, it is that all the technical points discussed above are key for BCA's WTO consistency.

1.4.1 World Trade Organization principles

The WTO was created in order to promote free trade by prohibiting unjustified protection and discrimination. The legal principle underlying all WTO regulation is the non-discrimination principle, divided into two key principles: the National Treatment principle (NT, article I) and the Most Favoured Nation principle (MFN, article III). NT prohibits country A to discriminate against country B or country C products over its own goods, whereas MFN forbids country A to discriminate against country B goods over country C goods (Avner, 2007).

BCA could then respect the general regime of WTO providing they respect these core principles. However, a second-best option could be to fall under the GATT exception regime (article XX). Indeed, providing they are not used as a means of arbitrary discrimination (article XX chapeau, which is a lighter version of art. III), measures that do not find justification under the general regime can still be implemented if they follow one of the eight subparagraphs of art. XX. In the case of BCA, it could be Art. XX (b) or (g), if BCA are considered "necessary to protect human, animal, or plant health of life" or "relating to the conservation of natural resources".

In practice, assessing whether a version of BCA may follow the general or the exception regime of WTO involves answering many technical questions that are beyond the scope of this chapter. To convey a glimpse of the type of legal reasoning, this section briefly discusses perhaps one of the most important questions: can two products that differ onlyin their carbon content be considered "unlike" products? If the answer is positive, the discrimination between these two products under BCA does not violate the MFN principle. A difference in carbon content for "same" products is called, in WTO technical language, a difference in PPM (Product and Production Method, basically the way products are made). WTO distinguishes PPM into two categories: product-related PPM and non-product-related PPM, whether the PPM is considered "incorporated in the product" or not. First, legal experts disagree on whether carbon emissions are a product-related or a non-product-related PPM, depending on the interpretation of "incorporated in the product", whether as "physically present in the product" or "part of the product". Second, WTO rules allow discriminating product-related PPM, but are unclear for non-product-related PPM. A conservative interpretation would say that products differing only in non-product-related PPM are “like” products, but recent case law seems to take a different direction (Low et al., 2011).

1.4.2 Lessons from the past

Sections 1.4.2 and 1.4.2 briefly explain two cases that provide some insights into the hypothetical consequences of BCA implementation, the first (the shrimp-turtle dispute) on the legal side and the second (the aviation inclusion in the EU ETS) on the political side.

The Shrimp-Turtle dispute

In order to protect five endangered species of sea turtles, the US banned in 1989 shrimps coming from countries where shrimpers where not equipped with turtle-excluders devices, a compulsory measure for US shrimp trawlers. In 1997, a coalition composed of India, Malaysia, Pakistan and Thailand challenged the US under the WTO, arguing that the import prohibition (Section 609 of Public Law 101-162) was inconsistent with the WTO rules. The Dispute Settlement Panel gave reason to the coalition, both in first judgement and in appeal in 1998. The main reason was that the embargo undermined members’ autonomy to determine their own policies, because it focused on turtle-excluder devices and did not provide enough flexibility in turtle protection policies to third countries. After this dispute the US revised the conditions of Section 609. But these were still not satisfactory for Malaysia, which challenged again the US in 2000. The WTO this time gave reason to the US both in the first judgement and in appeal in 2001. It founded that US provided “good faith” in negotiating an international agreement on the protection and conservation of sea turtles, as it was recommended by the Appellate Body. It also concluded that conditioning market access on the adoption of a programme comparable in effectiveness allowed for sufficient flexibility.

The political ordeal of aviation inclusion in the EU ETS

On November 2008, Directive 2008/101/EC launched the inclusion of aviation in the EU ETS starting in January 2012. Most of the allowances were supposed to be freely distributed, but because of the decrease of the global cap, the expected growth in air traffic and the limited ways of mitigation, it became clear that airline companies were going to buy a growing number of credits over time for their compliance. Despite some precautions (free allowances, sophisticated rules protecting fast-growing companies, use of revenues for climate-related initiatives, exemptions in case of "equivalent measures" in other countries), the European Commission has received a series of attacks coming from airline companies, their trade bodies and governments. The points at issue were sovereignty, the Common but Differentiated Responsibilities principle of the UNFCCC (as airlines from developing and developed countries received the same treatment), the Chicago Convention of 1944 limiting taxation on aviation commercial fuel, and the use of revenues (Sandbag, 2012). In 2012, a growing "coalition of the unwilling" led by China, India, Russia and the US agreed a series of retaliatory measures if EU states imposed sanctions for non-compliance. These pressures led the European Commission to "stop the clock" in November 2012, proposing a one-year deferring of the application of the scheme for intercontinental flights, leaving time to ICAO, the International Civil Aviation Organisation, to adopt a global policy. The implementation of the scheme as proposed in the directive remains highly uncertain at this time.

Conclusion

The shrimp-turtle case teaches us that the exception regime of the WTO can rule, that this institution takes seriously into account the attempt to conclude international agreements before implementing trade measures (Tamiotti, 2011), and that flexibility was the cornerstone of WTO dispute panel decisions (Zhang, 2012). However the degree of legal complexity of BCA is far beyond a simple ban on shrimps.

The setbacks of the inclusion of aviation in the EU ETS show us that countries are deeply reluctant to relinquish some of their sovereignty, especially when financial consequences are at stake. One can reasonably assume that BCA for EITE industries are more controversial in terms of political acceptance than the inclusion of aviation in the EU ETS. Then, BCA implementation would certainly involve a strong diplomatic and economic response, especially from the developing countries.

1.4.3 Political and legal challenges of Border Carbon Adjustments

International institutions state that free trade has a role to play in climate policies by promoting clean technology transfer and suppressing murky subventions to dirty sectors, but remain ambiguous concerning the legality of BCA (Bank and UNEP, 2007; Olhoff et al., 2009). The joint UNEP-WTO report (2009, p. 89) reads: "the general approach under WTO rules has been to acknowledge that some degree of trade restriction may be necessary to achieve certain policy objectives, as long as a number of carefully crafted conditions are respected". Legal experts are also divided on the subject, the bottom line of most analyses is that legal acceptability and political feasibility of BCA would depend on the specific designs of such measures (Tamiotti, 2011). There is no guarantee of the legal success and political acceptability of BCA, but two features would help. First, in-depth discussions with third countries to identify the potential points of conflict, rather than unilateral imposition of trade measures, are desirable (Low et al., 2011)). Second, flexibility must be a central piece of the policy package, which could mean allowing third countries national "comparable action" instead of systematic border carbon pricing.

Even with all these legal precautions, one can reasonably assume that, if BCA were to be implemented, third countries would publically condemn it as "green protectionism" or "eco-imperialism" (Dröge, 2011). WTO and UNFCCC share the unpleasant fact of being bogged down in international negotiations blockage (the next step of the Kyoto Protocol for UNFCCC, and the Doha round for WTO), and a clash between climate and trade regimes would be detrimental to both global trade and climate agreements.

If BCA are not likely to be implemented in the following years, they will undoubtedly be considered more and more, as abatement targets gaps are growing among countries. A "weak" version of BCA, based on best available technologies benchmark with the handing back of revenues, would seem the most preferable option, offering less vulnerability to a potential WTO dispute and giving certain compensations to other countries (Godard, 2009; Ismer and Neuhoff, 2007).

1.5 Conclusion

The reality for the foreseeable future is that climate policies will remain sub-global. Different mitigation targets among countries are legitimate under the Principle of Common but Differentiated Responsibilities (Zhang, 2012), but too uneven climate policies are less efficient if they cause carbon leakage and are unlikely to survive the national policy-making process if they entail significant competitiveness losses. These concerns are among the main arguments against the implementation of stringent climate policies in industrialized countries. How worrying are they?

Ex post studies have not shown significant evidence of leakage to date, but arguably the climate policies implemented so far may have been too moderate to allow measurement ofsuch effects. Ex ante studies indicate a leakage in the range of 5 to 20% in case of unilateral climate policies without measures to mitigate leakage. However, the induced diffusion of climate-friendly innovations generates abatement even in regions without climate policies, which may well compensate for leakage. Thus, leakage is clearly not a convincing argument against climate policies, although it invites actions to complement carbon pricing with specific measures in order to maximise their efficiency. Is competitiveness a more convincing argument against climate policies? Carbon costs matter, but they are one factor out of many (capital abundance, labour force qualification, proximity to customers, infrastructure quality, etc.) contributing to the competitiveness of an industry (Monjon and Hanoteau, 2007). Massive environmental relocations in case of stringent policies announced by Energy Intensive Trade Exposed (EITE) trade associations are not realistic: because these industries are very capital-intensive, they are less prone to relocation in general compared to "footloose" industries (Ederington et al., 2003). In the case of the EU ETS, competitiveness concerns have led to an over-allocation of permits, a generous use of offsets from the CDM and JI and finally a crash in carbon price. At this time the European Commission is struggling to tackle the growing structural supply-demand imbalance. The modest proposition of back-loading 900 million of allowances was rejected on 16 April

2013 by the European parliament, mainly for competitiveness reasons.[6] Hence, competitiveness, which was called a "dangerous obsession" for macroeconomic policy by Krugman (1994), may be so for climate policy as well. That said, because of the influence of EITE industries in the policy process, specific measures to protect these sectors are part of every realistic policy package.

Moreover, they may allow countries in the abating coalition to raise the ambition of their climate policy, and also extend the size of the climate coalition, as they would lessen the incentives of free-riding. Simply exempting these sectors is too costly to be justifiable: since emissions in these sectors would not be reduced, more abatement should take place in the others, including less cost-effective options. On purely economic grounds and from the point of view of the abating coalition, economic analysis favours the implementation of BCA, but from a legal and diplomatic point of view, the situation is much less clear-cut. If properly discussed with emerging economies, a BCA based on best available technology benchmarks, with revenues earmarked for climate-related projects in developing countries, may be the best solution. A fall-back option is to distribute free allowances in proportion to current output of EITE industries (output-based allocation): although less cost-effective, it could be an acceptable compromise between efficiency and feasibility. However, just as free allowances based on historic or capacities, the option implemented in the EU ETS, it could generate massive lobbying and competitive distortions since every industry tries to receive as much allowances as possible. Besides, the WTO compatibility of output-based allocation is not more granted than that of BCA (James, 2012).

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[6]The spokesman for Conservative MEPs declared: " We fear [backloading] will (...) encourage further carbon leakage, and undermine much-needed market predictability as the EU economy strives to find a way out of the economic crisis" (source: `http://www.guardian.co.uk/environment/2013/apr/16/meps-reject-reform-emissions-trading`), arguments mainly taken from the position of the Alliance of Energy Intensive Industries (source: `http://www.cembureau.be/sites/default/files/documents/AEII`).

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2

Would border carbon adjustments prevent carbon leakage and heavy industry competitiveness losses? Insights from a meta-analysis of recent economic studies

A growing body of academic literature has been developed in the recent years to quantify the impacts of uneven climate policies on carbon leakage and competitiveness, and to find the best policy measures to counteract them. Among them, border carbon adjustments (BCAs), which consist in taxing products at the border on their carbon content, are widely discussed. Some literature reviews have been published recently synthesizing these studies (Chapter 1 and Zhang (2012); Quirion (2010); Dröge (2009); Gerlagh and Kuik (2007)) but to our knowledge no quantitative meta-analysis has been conducted.

Meta-analysis is a method developed to provide a summary of empirical results from different studies and test hypotheses regarding the determinants of these estimates (Nelson and Kennedy, 2009). It has been extensively used in medical research. The first meta-analysis in economics can be traced back to Stanley and Jarrell (1989). In the field of environmental and resource economics, the majority of meta-analyses summarizes the results of different nonmarket valuation studies (Van Houtven et al., 2007; Brander and Koetse, 2011; Barrio and Loureiro, 2010; Ojea and Loureiro, 2011; Richardson and Loomis, 2009). Closer to our subject, one can cite two studies on marginal abatement costs to mitigate climate change, one for all sectors (Kuik et al., 2009) and the other specific to agriculture (Vermont and De Cara, 2010). An extensive review of meta-analysis methods in environmental economics is given in Nelson and Kennedy (Nelson and Kennedy, 2009).

In this chapter, we conduct a meta-analysis on 25 studies dating from 2004 to 2012, altogether providing 310 estimates of carbon leakage ratios according to different assumptions and models. The typical range of carbon leakage estimates is from 5% to 25% (mean 14%) without policy and from -5% to 15% (mean 6%) with BCAs. We conduct a meta-regression analysis to further investigate the impact of different assumptions on carbon leakage estimates. Impact of key model parameters, such as Armington elasticities, and policy features such as linking carbon markets or extending pricing to all greenhouse gases sources can be highlighted. We find that, all other parameters being constant, BCAs implementation reduces the leakage ratio by 6 percentage points.

The remainder of this chapter is structured as follow. Section 2.1 describes the database and section 2.2 provides some descriptive statistics. The meta-regression model is explained in section 2.3 and results are discussed in section 2.4. Section 2.5 concludes.

2.1 Database description

Many articles and working papers deal with carbon leakage and competitiveness issues but only some of them are models giving *ex ante* numerical estimates. The body of literature regarding these issues also comprises *ex post* econometrical analyses, analytical models and political or juridical studies (Cosbey et al., 2012; Ismer and Neuhoff, 2007; Monjon and Quirion, 2011b). The first criterion to be part of our sample was to provide numerical estimations of carbon leakage with a model. The second criterion was, since the purpose of this chapter is to investigate the impact of border carbon adjustments on leakage, to include BCAs in the scenarios. Thirdly, we discarded old studies (before 2004) to focus on the recent literature.

To constitute our sample, we searched for studies in standard search engines (Web of Science, Google Scholar) and cross references with keywords "carbon leakage" and "border carbon adjustments". The research was completed in December 2012. Our sample is made of 25 studies dating from 2004 to 2012, most of them (14) are part of the recent Energy Economics Special Issue. Some are grey literature (MIT working paper, World Bank working paper, etc), others are published in energy economics and environmental economics journals (Energy Economics, Energy Policy, the Energy Journal, Energy Policy, Climate Policy etc). The majority are computable general equilibrium (CGE) models which rely on the GTAP database (except for one), the others are sectoral or multi-sectoral partial equilibrium models. The number of carbon leakage estimates per study varies from 2 (Weitzel et al., 2012) to 54 (Alexeeva-Talebi et al., 2012), with a mean of 12.6.

The studied effect-size in the meta-regression analysis is the leakage-to-reduction ratio or leakage ratio,

$$l = \frac{\triangle E_{NonCOA}}{-\triangle E_{COA}}$$

where $\triangle E_{COA}$ is the emissions variation in the climate coalition between the climate policy scenario and the counterfactual business-as-usual scenario, and $\triangle E_{nonCOA}$ the emissions variation in the rest of the world. Its common use avoids us to make approximate conversions between studies. In other words all studies calculate the same thing, which is necessary in a meta-analysis as a "synthesis requires the ability to define a common concept to be measured"(Smith and Pattanayak, 2002)).

In the majority of the cases results were available on tables, but sometimes they were taken from graphs or derived from own calculation like in Mattoo et al. (2009).

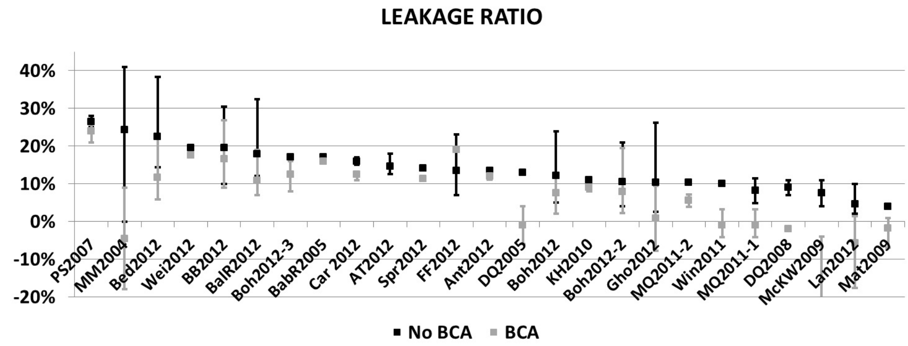

Figure 2.1: Leakage ratio in selected studies (mean, minimum and maximal values with or without BCAs), ranked by mean value without BCAs

2.2 Descriptive Statistics

2.2.1 First sight

Figure 2.1 presents ranges of leakage ratio estimates for the 25 studies (mean, minimum and maximal values with or without BCAs). Leakage ratio estimates range from 2% to 41% without BCAs and from -41% to 27% with BCAs. Eight studies find negative values of leakage ratio in case of BCAs, with three studies (Mathiesen and Maestad, 2004; McKibbin et al., 2008; Lanzi et al., 2012) finding values below -15%. Internal variations (within one study) of leakage ratio estimates range from almost null (Alexeeva-Talebi et al., 2007) to relatively high (Mathiesen and Maestad, 2004; Bednar-Friedl et al., 2012; Ghosh et al., 2012) depending on the scenarios and models.

Comparing scenarios by pair (with and without BCAs, all the other parameters being constant), we can observe that in all cases, BCAs led to a reduction of the leakage ratio.[1] These results are in contrast with (Jakob et al., 2013) who found that BCAs could increase the leakage ratio[2]. For each pair, we calculate the

[1] In figure 2.1, for FF2012 (Fischer and Fox, 2012), the mean with BCAs is higher than with no BCAs, but the "equivalent" BCAs scenarios corresponds to the highest value of leakage ratio of the no BCAs scenario (Europe only abating).

[2] In this paper, under certain conditions, if in non-coalition countries, the carbon intensity of exports ("clean" sector) is higher than those of local production ("dirty" sector), a reallocation of production induced by BCAs from "clean" to "dirty" sector would increase emissions in non-coalition countries and then leakage ratio on a global scale.

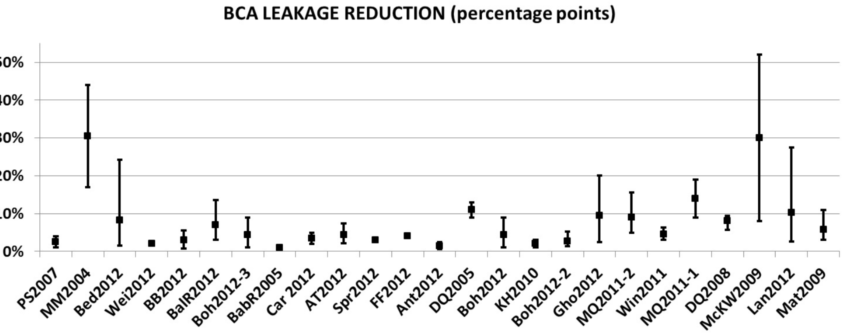

Figure 2.2: Leakage ratio reduction in case of Border Carbon Adjustment (same ranking as in figure 2.1)

leakage ratio reduction in percentage points (defined as $LeakageRatioReduction = LeakageRatio_{NoBCAs} - LeakageRatio_{BCAs}$). In the majority of the cases, the leakage ratio reduction due to BCAs stands between 1 and 15 percentage points, but there are some outliers above 30 percentage points, where BCAs actually generates negative leakage ratios (McKibbin et al., 2008; Mathiesen and Maestad, 2004).

Apart from carbon leakage, competitiveness losses in energy-intensive industries constitutes the other component of the climate trade nexus. Though extensively used in the public debate, the notion of competitiveness remains ambiguous (Alexeeva-Talebi et al., 2007). Some authors consider that this notion is meaningless at the national level (Krugman, 1994). At the sectoral level, it may refer to "ability to sell" or "ability to earn". In CGE models, competitiveness is most of the time implicitly defined as "ability to sell" and measured by gross output. In our sample, 17 of the 25 studies show results of output change for industries. Based on GTAP sectors, EITE sectors often regroups refined goods, chemical products, non-metallic minerals, iron and steel industry and non-ferrous metals (although sometimes refined goods is aside). Some studies present only disaggregated results by sectors, and not the output change for EITE sectors as a whole. In this case, we use the average of the output of iron and steel and non-metallic minerals sectors (or average of cement and iron and steel) as a proxy for EITE sectors.[3]

The output change of EITE sectors varies from -0.1% to -16% without BCAs

[3]For the only two studies where output changes were available both by sector and for EITE sectors as a whole (Lanzi et al., 2012; Ghosh et al., 2012), it was a correct proxy. Iron and Steel (resp. Non-Metallic Minerals) being a bit less (resp. more) impacted than EITE as a whole.

and from +2.2% to -15.5% with BCAs. There is a clear dichotomy between CGE models where output loss range is 0%-3% (except for Alexeeva-Talebi 2012 (b) and Ghosh et al. 2012 where it is a bit more (around 3%-7%)) and sectoral partial equilibrium models where output loss range is 8%-15%. In all cases, BCAs reduce the output loss among EITE industries[4] and in five cases (Peterson and Schleich, 2007; Alexeeva-Talebi et al., 2007; Kuik and Hofkes, 2010; Mattoo et al., 2009; Ghosh et al., 2012), the output variation of EITE industries is even positive.

The welfare (or in some studies GDP) variation of the abating coalition ranges from -1.58% to 0.02% without BCAs and from -0.9% to 0.40% with BCAs (the environmental impact is never taken into account in the welfare estimation[5]). Though BCAs improve welfare of coalition countries compared to a no BCAs scenario, they most of the time do not reestablish a "neutral" situation (e.g a variation near 0%). The welfare variation is still negative after BCAs, because the consumers of the coalition pay higher prices in EITE sectors' products. This improvement of welfare in coalition countries goes hand in hand with a degradation of welfare in non-coalition countries. BCAs have big distributionnal impacts: they transfer a part of the burden to the non-coalition countries (Böhringer et al., 2012c). In the studies that report it (Böhringer et al., 2012c; Lanzi et al., 2012; Mattoo et al., 2009), *global* welfare is decreasing with BCAs.

2.2.2 Merging studies

Gathering all the estimates of carbon leakage in the 25 studies, we compute kernel density estimations for the estimates accross all studies. As the number of estimates varies greatly (from 2 to 54) across studies, we consider two ways of merging results, the "scenarios equality" method and the "articles equality" method. In the "scenarios equality" method, we add all estimates regardless of the article they are from. Then an article with N estimates "weights" $N/2$ times more in the final distribution than an article with only two estimates. In the "articles equality" method however, weights are put on estimates to assure that each article "weights"

[4]However in the CASE model (Monjon and Quirion, 2011a,b), cement output is more reduced in the presence of BCAs.

[5]In the Energy Economics special issue, leakage is endogenously compensated by a higher abatement to assure a same environmental impact in all scenarios in order to compare welfare variations.

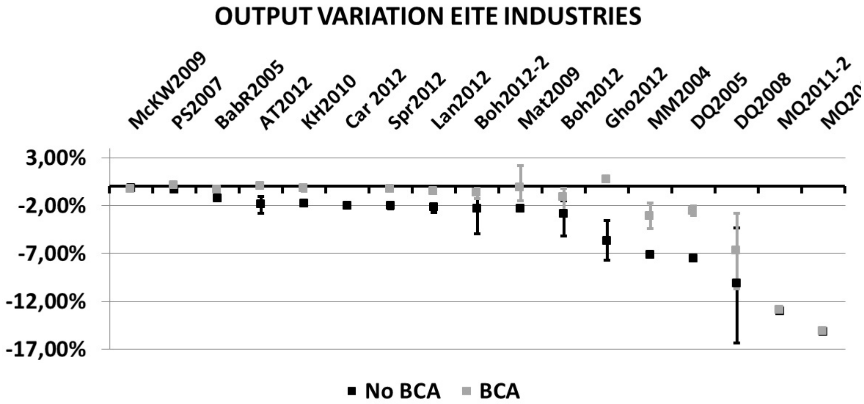

Figure 2.3: Output change of EITE industries in selected studies (ranked by mean value without BCAs)

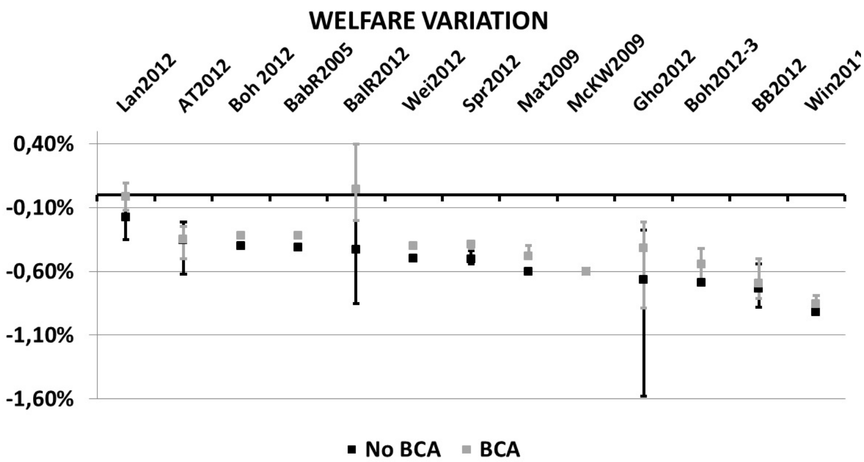

Figure 2.4: Welfare variation in abating coalition (ranked by mean value without BCAs)

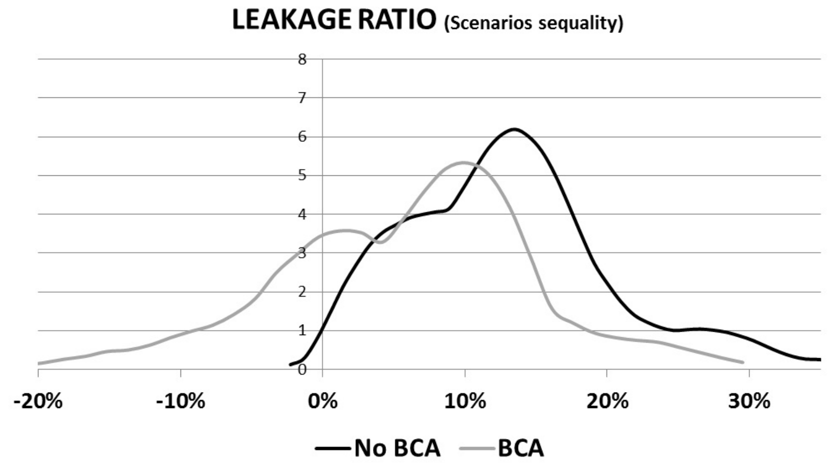

Figure 2.5: Leakage Ratio. "scenarios equality" merging

the same in the final distribution.[6] By this process the distribution of results with the "articles equality" method is less smooth because there are artificially some accumulation in the distribution. However the distributions share the same shape with both results, especially for the leakage ratio and the output variation of EITE industries, which can be interpreted as a sign of the robustness of the results.

Both leakage ratio distribution and EITE output change distribution are bimodal. For leakage ratio without BCAs there is a concentration around 5% and another around 12%.[7] We can see that a leakage ratio above 100%, theoretically possible if the carbon content of products is higher outside the climate coalition is well out of the range of estimates in the literature. For EITE output variation there is a concentration at -2% and another one (more spread out) at -7%, which can be interpreted as the dichotomy between CGE models and PE models. The coalition welfare variation distribution is unimodal, with a mode of -0.6% without BCAs and -0.3% with BCAs.

One can easily visualize in figures 5 to 10 the impact of BCAs in reducing the leakage ratio, restoring some competitiveness and to a lesser extent improving

[6] If N_k is the number of estimates in the article k, the weight for an estimate from article i is then $\frac{\max_k(N_k)}{N_i}$ (and the closest integer value for kernel estimate using Stata). In this case each article weights $\max_k(N_k)$ in the final distribution.

[7] Not a single estimate of leakage ratio is negative without BCAs, the negative part is an artifact in the kernel density estimation

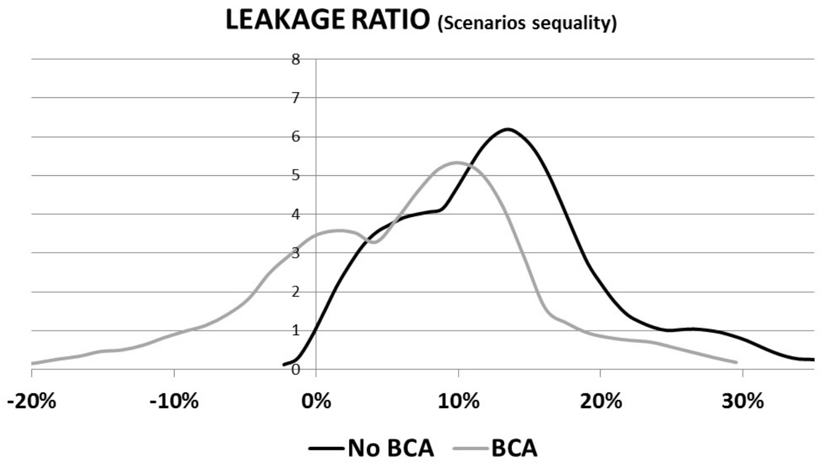

Figure 2.6: Leakage Ratio. (Kernel density estimates)

coalition welfare with the left shift of the leakage ratio distribution and the right shifts of output change and coalition welfare change distributions.

2.3 Meta-regression analysis

2.3.1 Methods

Meta-regression is widely used in meta-analysis as it is an interesting way to go beyond standard literature review, by combining numerical results from different studies in a statistical manner (Vermont and De Cara, 2010; Kuik et al., 2009; Horváthová, 2010). Guidelines on how meta-regression analysis of economics research should be conducted were recently published (Stanley et al., 2013). The guidelines were divided into three topics: research questions and effect size; research literature searching, compilation and coding; and meta-regression modeling issues.

The first topic is discussed in the introduction (general context of the research question and statement of the effect studied) and in the end of part 2.1 (how the effect size is measured by the leakage ratio, which is a common metric). The second topic is discussed in the beginning of part 2.1 (how the literature was searched and what are the criteria for study inclusion). Table 2.1 gives detailed information on the articles used in the meta-analysis. Stanley et al. (2013) encourage that two

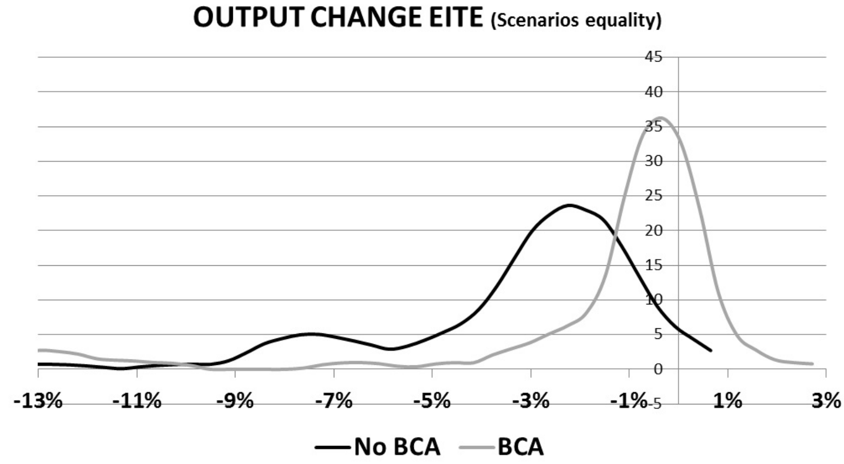

Figure 2.7: Output change of EITE industries (Kernel density estimates). "scenarios equality" merging

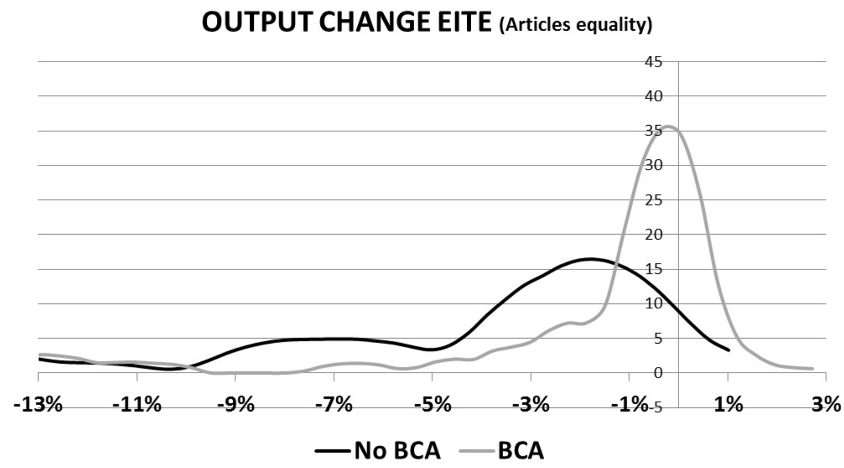

Figure 2.8: Output change of EITE industries (Kernel density estimates). "articles equality" merging

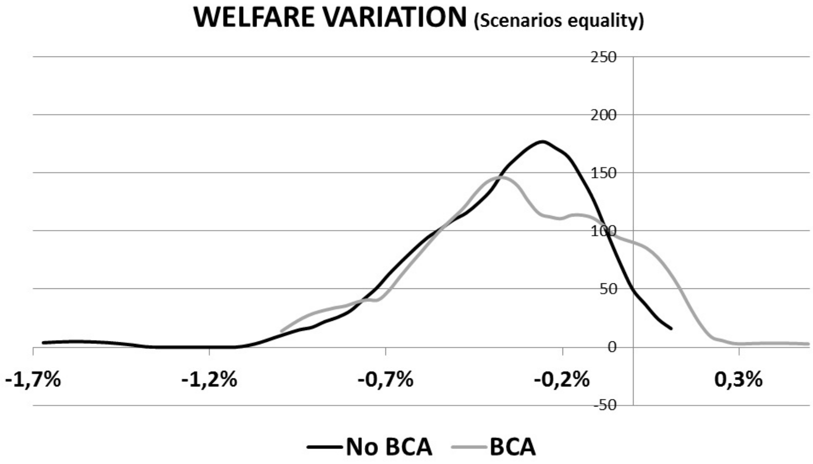

Figure 2.9: Welfare variation (Kernel density estimates). "scenarios equality" merging

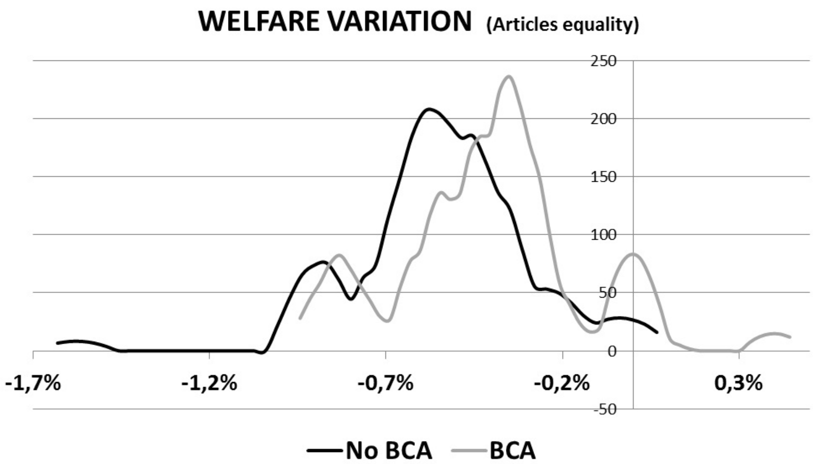

Figure 2.10: Welfare variation (Kernel density estimates). "articles equality" merging

Table 2.1: Selected studies

Name	Reference	Journal	Model Name	Model type	Main Database	Cluster	Obs[†]
Boh2012	Böhringer et al. (2012a)	En Eco (SI)[1]	Several	CGE	GTAP 7.1	1	26 (14+12)
Gho2012	Ghosh et al. (2012)	En Eco (SI)	EC-MS-MR	CGE	GTAP 7.1	7	18 (6+12)
AT2012	Alexeeva-Talebi et al. (2012)	En Eco (SI)	PACE	CGE	GTAP 7.1	2	54 (27+27)
Lan2012	Lanzi et al. (2012)	En Eco (SI)	ENV-Linkages	CGE	GTAP 7.0	3	44 (20+24)
Boh2012-2	Böhringer et al. (2012b)	En Eco (SI)	BCR	CGE	GTAP 7.1	8	18 (9+9)
BalR2012	Balistreri and Rutherford (2012)	En Eco (SI)	MINES	CGE	GTAP 7.0	13	10 (5+5)
Wei2012	Weitzel et al. (2012)	En Eco (SI)	DART	CGE	GTAP 7.0	13	2 (1+1)
FF2012	Fischer and Fox (2012)	En Eco (SI)	GTAPinGAMS	CGE	GTAP 7.0	10	5 (4+1)
BB2012	Boeters and Bollen (2012)	En Eco (SI)	WorldScan	CGE	GTAP 7.0	14	9 (3+6)
Spr2012	Springmann (2012)	En Eco (SI)	CVO	CGE	GTAP 7.1	11	7 (4+3)
Car 2012	Caron (2012)	En Eco (SI)	CEPE	CGE	GTAP 7.0	11	8 (4+4)
Bed2012	Bednar-Friedl et al. (2012) et al.	En Eco (SI)	WEG_CENTER	CGE	GTAP 7.0	4	24 (12+12)
Boh2012-3	Böhringer et al. (2012c)	En Eco (SI)	SNOW	CGE	GTAP 7.1	12	10 (1+9)
Ant2012	Antimiani et al. (2012)	En Eco (SI)	GTAP-E	CGE	GTAP 7.1	10	3 (1+2)
Mat2009	Mattoo et al. (2009)	World Bank WP[2]	ENVISAGE	CGE	GTAP 7.0	9	6 (1+5)
McKW2009	McKibbin et al. (2008)	Lowy Institute WP	G-Cubed	CGE	n.a	9	4 (2+2)
PS2007	Peterson and Schleich (2007)	ISI Working Paper	GTAP-E	CGE	GTAP 6.0	15	6 (2+4)
KH2010	Kuik and Hofkes (2010)	Energy Policy	GTAP-E	CGE	GTAP 6.0	10	3 (1+2)
Win2011	Winchester et al. (2011)	MIT Working Paper	EPPA	CGE	GTAP 6.0	9	5 (1+4)
BabR2005	Babiker (2005)	The Energy Journal	No Name	CGE	GTAP 5.0	12	2 (1+1)
MM2004	Mathiesen and Maestad (2004)	The Energy Journal	SIM	PE		6	11 (9+2)
MQ2011-1	Monjon and Quirion (2011a)	Ecological Economics	CASE II	PE		5	20 (10+10)
DQ2005	Demailly and Quirion (2005)	OECD Report	CEMSIM-GEO	PE		6	3 (1+2)
DQ2008	Demailly and Quirion (2008)	Energy Economics	CASE I	PE		6	6 (3+3)
MQ2011-2	Monjon and Quirion (2011b)	Climate Policy	CASE II	PE		5	6 (2+4)

[†] The numbers in parenthesis detail the number of leakage ratio estimates without and with BCAs implementation
[1] Energy Economics Special Issue [2] WP=Working Paper

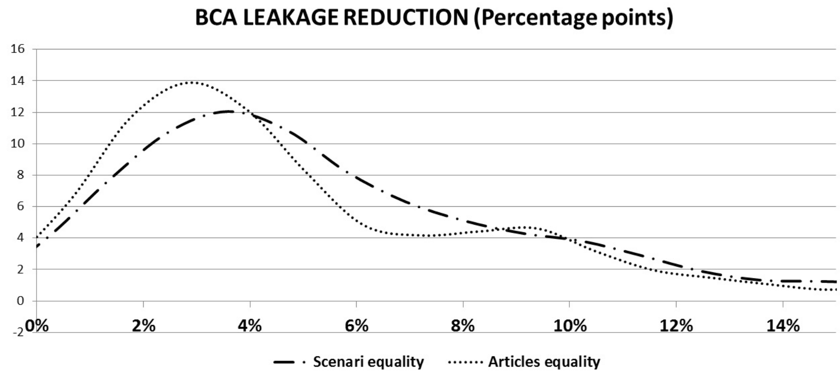

Figure 2.11: BCAs leakage reduction (in percentage points)

or more reviewers should code the relevant research. In this study, only the first author searched and coded the research literature.

In the rest of this part, we will detail the third topic (modeling issues). As recommended as good practice, we display descriptive statistics of the variables that are coded (see Table 2.7 in the appendix 2.6.2 for the effect size e.g. the leakage ratio, and Table 2.2 for the regression variables). Publication bias is a major issue in meta-regression analysis. This form of sample selection bias occurs if primary studies with statistically weak or unusual results are less likely to be published (Nelson and Kennedy, 2009). For example, it has been widely recognized to exaggerate the effectiveness of pharmaceuticals (Doucouliagos and Stanley, 2009). Statistical techniques to take this bias into account exist (Stanley, 2005; Rothstein et al., 2006; Havranek, 2013) but they require standard errors of the estimates. We cannot apply these methods here since we deal with numerical studies (no statistical significancy is involved and then no standard errors are given with the results). It is highly likely that a publication bias also exists in the area of modeling studies: authors compare their results with those of the literature and are able to change the settings or calibration of their models to influence the results. Our best option to address this issue was to embrace as many studies as possible without artificially setting aside some of them, e.g. non peer-reviewed papers.

Another potential issue is the existence and the treatment of outlier observations (some estimates that are unrepresentative or overly influential). To discard outliers we first perform a robust estimation procedure using iteratively Huber weights[8] (Huber, 1964) and only keep estimates whose final weights are above a certain threshold (5%). Among the first original samples of 310, 144 and 166 values (for "All", "no BCAs" and "BCAs", see later), 16, 10 and 6 observations were dismissed (20 out of 25 articles have less than one discarded estimate). The fact that the articles with the more discarded estimates were the relatively old ones (Mathiesen and Maestad (2004), Babiker (2005) and Peterson and Schleich (2007)) suggests that the literature is converging (which could also reflect a publication bias...). Estimation of parameters without the treatment of outliers is given in the appendix 2.6.1 as a sensitivity analysis. The reader can verify that the exclusion of outliers slightly improves the statistical significance of some coefficients without substantially affecting the results of the meta-regression analysis.

Finally, heteroskedasticity in effect size variance and non-independence of observations of the same primary studies due to within study dependence has long been recognized as a potential estimation problem for meta-regression (Nelson and Kennedy, 2009). Some authors favor the use of a "best-set" of estimates, meaning a single estimate per study (Stanley, 2001) but this shrinks dramatically the pool of estimates. In our case we used a Random Effect Multi-Level (REML) model with study identifiers as in Doucouliagos and Stanley (2009), to control for the potential dependence of estimates within a primary study. A second method, a "cluster-robust" OLS estimator,[9] as in Kuik et al. (2009) and Vermont and De Cara (2010)), is used as a sensitivity analysis (see appendix 2.6.1). Some differences on the coefficient values exist but overall our findings are robust to the method employed.

2.3.2 The model

To investigate the sources of heterogeneity among the different carbon leakage estimates, we test three variations of the meta-regression model based on different samples: one for all leakage ratio estimates, one for estimates in the absence of

[8]The Stata command that is used is rreg.

[9]The observations are gathered in 15 clusters (see Table 2.1). Studies with many observations are the first clusters (with the exception of Monjon and Quirion (2011a) and Monjon and Quirion (2011b) which are merged because results are from the same model CASE II), then studies that share common features are gathered in same clusters (2 or 3 studies per cluster representing 10-15 observations).

BCAs and the last one for estimates in the presence of BCAs:

$$\begin{aligned} Leakage_{ij} = {} & Const + \beta_1 GE_{ij} + \beta_2 Coasize_{ij} + \beta_3 Abatement_{ij} \\ & + \beta_4 Link_{ij} + \beta_5 GHG_{ij} + \beta_6 Armington_{ij} \\ & + \beta_7 BCAs_{ij} + u_{ij} \end{aligned} \tag{2.1}$$

$$\begin{aligned} Leakage_{NoBCAs,ij} = {} & Const + \beta_1 GE_{ij} + \beta_2 Coasize_{ij} + \beta_3 Abatement_{ij} \\ & + \beta_4 Link_{ij} + \beta_5 GHG_{ij} + \beta_6 Armington_{ij} \\ & + u_{ij} \end{aligned} \tag{2.2}$$

$$\begin{aligned} Leakage_{BCAs,ij} = {} & Const + \beta_1 GE_{ij} + \beta_2 Coasize_{ij} + \beta_3 Abatement_{ij} \\ & + \beta_4 Link_{ij} + \beta_5 GHG_{ij} + \beta_6 Armington_{ij} \\ & + \beta_8 Exp_{ij} + \beta_9 Foreign_{ij} + \beta_{10} AllSect_{ij} + \beta_{11} Indirect_{ij} \\ & + u_{ij} \end{aligned} \tag{2.3}$$

where $Leakage_{ij}$ is the i-th estimate of leakage ratio reported in the j-th study.

The choices of the variables in the models are driven by the scenarios and the available data in the studies, as well as the debates in the literature. The first variables are *GE* (a dummy variable set equal to 1 if the model is a CGE), *Coasize* (the size of the abating coalition in percentage of worldwide emissions[10]) and *Abatement* (the abatement target)[11]. Then we have two dummies related to scenarios *Link* (if permit trading is authorized between the different regions of the coalition[12]) and *GHG* (if all carbon sources, and not only CO2 are considered).

Armington elasticities, which are used to model international trade, are con-

[10] In the overwhelming majority of the articles, the coalitions were centered on Europe, in several cases enlarge to Annex 1 except Russia (A1xR) or A1xR plus China. Therefore no variable was considered to describe the coalition in itself (for example EU or US), but only its size in terms of worlwide emissions.

[11] The logarithm of *Coasize* and *Abatement* have been tried as variables without changing the statistical significancy of the results

[12] which supposes that the abating coalition is composed of more than one region in the model. For example if Europe is the abating coalition it is not considered that permit trading is allowed.

sidered as a crucial parameter in leakage ratio estimates (Monjon and Quirion, 2011a; Alexeeva-Talebi et al., 2012; Balistreri and Rutherford, 2012). Most of the time they were not explicitly displayed in the articles. However some studies made sensitivity analyses on this parameter (for example doubling or dividing in half the original values). In the meta-analysis, the *Armington* parameter is then, rather than a numerical value, an "almost dummy" linked with "high" (+1), "low" (-1), "very high" (+2) or "very low" (-2) Armington elasticities values[13] when sensitivity analysis were performed on these parameters. It would have been interesting to incorporate a parameter for the fossil fuel supply elasticity which is also recognized to be determinant in the leakage ratio estimations for the international fossil fuel channel (Light et al., 1999; Gerlagh and Kuik, 2007). However, because they were not available most of the time, it was decided not to take them into account in the meta-regression.

BCAs is a dummy which takes the value of 1 if BCAs are implemented. It is the central parameter of our study since we primarily investigate to what extent BCAs are efficient to reduce leakage. Four dummies detail the policy features of the BCAs: *Exp* (if export rebates are part of the scheme), *Foreign* (if the adjustment is based on foreign specific emissions, instead of home specific emissions or best available technology), *AllSect* (if the adjustment concerns all sectors and not only EITE sectors), and *Indirect* (if indirect emissions are taken into account in the adjustment). Table 2.2 summarizes information about the regression variables.

2.4 Discussion of the results

Interpreting the results, one must bear in mind that, though meta-regression analysis is a powerful tool to incorporate all the sources of variability in a single model, one must interpret the results with caution. Indeed, the calculated coefficients depend not only on primary models, that made different assumptions, but also on the statistical variability of the parameters which is, except for the variable *BCAs*, far from being perfect. For example, *Abatement* is set at 20% for 61% of the cases and varies within three studies only (Böhringer et al., 2012a; McKibbin et al., 2008;

[13] In Balistreri and Rutherford (2012) the Melitz structure (Melitz, 2003) is considered equivalent to "very high" Armington.

Table 2.2: Meta-regression variables

Name	Variable type	Explanation	Summary statistics	Variability[1]	
GE	Dummy	1 if the model is a CGE	268 (87% of the cases)	Correct	0/20/5
Coasize	Percentage	Size of the abating coalition (percentage of worldwide emissions)	Mean 35% Mode: 15% (for 39% of the cases)	Good	9 - 8
Abatement	Percentage	Abatement target	Mean 19% Mode: 20% for 61% of the cases	Poor	4 - 8
Link	Dummy	Possibility to sell permits across the coalition	83 (27% of the cases)	Correct	2/11/12
GHG	Dummy	If carbon pricing is extended to all GHG sources	9 (3% of the cases)	Poor	2/23/0
Armington	5 values (-2/-1/0/1/2)	1 (resp. -1) corresponds to ”Armington high” (resp. ”Armington low”). 2 for ”Melitz” or higher Armington than ”Armington high”	31 for 1 and 37 for -1 (10% and 12% of the cases)	Correct	4 - 5
BCA	Dummy	1 if there is border carbon adjustments	167 (54% of the cases)	Very Good	25/0/0
Exp	Dummy	1 if export rebates are part of the scheme	146 (87% of the BCA cases)	Good	6/17/2
Foreign	Dummy	1 if the adjustment is based on foreign specific emissions (or average foreign). 0 if home (or BAT)	114 (68% of the BCA cases)	Good	8/9/8
Allsect	Dummy	1 if the adjustment concerns all sectors and not specifically Energy-intensive sectors	47 (28% of the BCA cases)	Good	4/6/15
Indirect	Dummy	1 if indirect emissions are taken into account in the adjustment	152 (91% of the BCA cases)	Poor	2/20/3

[1] For dummies, x/y/z means that the parameter takes both values 1 and 0 in x articles, 0 only in y articles and 1 only in z articles. For others, a-b means that the parameter takes more than two different values within a articles, and that there are b values taken by the parameter in total.

Demailly and Quirion, 2008). *Indirect* is set at the value 1 for 91% of the cases and varies within two studies only (Böhringer et al., 2012c; Monjon and Quirion, 2011b). This aspect is unavoidable in a meta-regression analysis as we take already made studies and do not design the scenarios by ourselves. We still include these "poorly variable" variables in the regression, and interpret the coefficients in the light of this aspect, knowing that they may be biased or may not appear as statistically significant as they may have been.

Table 2.3: Meta-regression results. REML estimation

	All	**No BCAs**	**BCAs**
GE	0.091***	0.047	0.124***
	(2.74)	(1.60)	(4.27)
Coasize	-0.214***	-0.221***	-0.147***
	(12.12)	(10.97)	(5.94)
Abatement	0.090	0.163*	0.084
	(1.04)	(1.78)	(0.69)
Link	0.003	-0.005	0.002
	(0.26)	(0.48)	(0.13)
GHG	-0.029**	-0.014	-0.062***
	(2.24)	(1.04)	(2.82)
Armington	0.019***	0.033***	0.003
	(4.68)	(7.75)	(0.51)
BCA	-0.063***		
	(14.27)		
Exp			-0.039***
			(2.98)
Foreign			-0.020*
			(1.90)
Allsect			-0.042***
			(2.90)
Indirect			-0.015
			(0.87)
N	294	134	160
Wald χ^2	386.13†	192.61†	78.25†
LR test	220.50†	96.95†	42.02†
DW test OLS	0.68	0.52	1.08

† $prob = 0.0000$

The results of the meta-regresssion are visible Table 2.3. We recall that three estimations are performed: one for all the leakage ratio estimates, one for those in the absence of BCAs and one for those in the presence of BCAs. The quality of the estimations is assessed through the Wald χ^2 test and the LR (Likelihood Ratio) test that compares the results with the linear regression. The presence of within-study dependence is revealed by the Durbin-Watson test (computed after a simple OLS estimation), and the LR test confirms that the use of a REML estimation is appropriate.

The difference between CGE models and other models is statistically significant and is positive (except for the "no BCAs" sample). We find that, all other parameters being constant, the leakage ratio estimate is 9 percentage points higher in CGE models and 12 percentage points in the case of BCAs implementation, which is a noteworthy difference. The lack of non-CGE models estimates (non CGE models constitute only one fifth of the articles and even less in terms of leakage ratio estimates) remains an impediment for the statistical value of this coefficient. An explanation could be that CGE models include both channels of leakage ratio, the competitiveness channel and the international fossil fuel channel, which is recognized to predominate (Gerlagh and Kuik, 2007; Fischer and Fox, 2012; Weitzel et al., 2012) whereas partial equilibrium models only include the first one (except for Mathiesen and Maestad (2004)).

The coefficient for the coalition size is negative and very statistically significant. Changing the size of the coalition from Europe (15% of world's emissions in 2004) to Annex 1 plus China except Russia (71% of world's emissions in 2004) would involve in the model a decrease of leakage ratio of about 12 percentage points without BCAs and 8 percentage points with BCAs.

Theoretically, the bigger is the abatement, the higher is the leakage in absolute terms (tons of carbon emissions). As the leakage ratio is the leakage in absolute terms divided by the abatement and the latter increases as well, there is an indeterminacy about the relationship between the abatement and the leakage ratio. In the meta-regression model, the correlation is positive, but the statistical significancy is weak (a p-value below 0.1 is reached only for the no-BCAs sample), which may be attributable to the small variability of this parameter. In Alexeeva-Talebi et al. (2007) (which was not included in our study because there was no BCAs), the correlation is negative (leakage of 32%, 29% and 27% for Europe abating respectively 10%, 20% and 30% of its emissions). In Böhringer et al. (2012b) however, the relationship is positive (leakage of 15.3%, 17.9% and 21% for Europe abating respectively 10%, 20% and 30% of its emissions).

Concerning the policy parameters, authorizing permit trading (linking) within the coalition is not statistically significant. In the two studies that change explicitly this parameter in the different scenarios (Lanzi et al., 2012; Springmann, 2012), permit trading diminishes leakage to a small extent. It is therefore the lack of variability *between studies* that may explain this non-significance (about half of the articles have permit trading in all their scenarios and the other half do not in all their scenarios).

Conversely, extending carbon pricing to all GHG sources is statistically significant, especially when BCAs are implemented (decreasing the leakage ratio by 6 percentage points). However the poor variability of this parameter diminishes the confidence we can grant to this econometric estimation (two articles study the coverage extension to all GHGs in one of their scenarios, but all the other articles only consider carbon emissions, i.e. there is no variability between studies).

The Armington parameter proves statistically significant (except for the "BCAs" sample) and is positive as expected. A higher value, meaning a more "flexible" international trade modeled, induces more impact of price differentiation across regions on trade flows, and therefore more leakage. In our meta-regression model, taking high values of Armington elasticities instead of low values would then lead to leakage ratio estimates about $2 \times 1.9 = 3.8$ percentage points higher.

With a very high p-value, we find that the *BCA* parameter is statistically significant and is negative. All other parameters being constant, BCAs implementation reduces the leakage ratio by 6 percentage points. This statistical finding fits the data in the descriptive statistics section (figure 2.11).

More specifically, among the BCAs options, export rebates and the inclusion of all sectors instead of only EITE sectors would have the most important impact (decrease of 4 percentage points of the leakage ratio for each), roughly the double than basing adjustment on foreign specific emissions instead of home specific emissions. In this meta-regression model it is not the politically and juridically risky option (foreign carbon content based adjustment) that would be the most efficient to reduce leakage but more an option with high administrative costs (adjustment to all sectors). The inclusion of indirect emissions is without surprise not statistically significant (there is very little statistical variability for this parameter). In the two studies where a change of this feature is included in the scenarios, it is proven to reduce leakage: in Böhringer et al. (2012c), from 0.5 to 2 percentage points, depending on the adjustment level, and in Monjon and Quirion (2011a), from 1.5 to 2 percentage points.

Meta-regression results can also be used to make out-of-sample predictions, which is called benefit transfer (Nelson and Kennedy, 2009; Van Houtven et al., 2007). This exercise is especially interesting for meta-analysis of empirical studies as they allow forecast for other locations or commodities which may save the employed resources to make additional surveys. Here we show as an example the results of leakage ratio estimations with the meta-regression analysis coefficients for different abating coalitions and policies (see Table 2.4). The estimated values of leakage ratio seem reasonable but the 95% confidence intervals are very wide.

2.5 Conclusion

A global climate policy is unlikely to be implemented in the years to come and the adoption of ambitious national or regional climate policies is hindered by claims of industry competitiveness losses and carbon leakage. Border Carbon Adjustment (BCAs) has been proposed to overcome these hurdles but its potential efficacy has been controversial. Moreover some authors argue that BCAs aims at protecting heavy industries competitiveness rather than at tackling leakage (Kuik and Hofkes, 2010) while other authors defend that BCAs implementation cannot be justified only for competitiveness motives (Cosbey et al., 2012). Finally, BCAs proposals differ by key design choices such as the inclusion of exports rebates, indirect (electricity-related) emissions, or the adjustment level, which can be the domestic or foreign average specific emissions, or best-available technologies. How BCAs performance would be impacted by these choices remains an open question.

To shed some light of these issues, we have gathered and analysed 310 estimates of carbon leakage and output loss in Energy-Intensive Trade-Exposed (EITE) sectors from 25 studies dating from 2004 to 2012. A meta-regression was conducted to capture the impact of different assumptions on the model results.

Across our studies, the leakage ratio ranges from 5% to 25% (mean 14%) without BCAs and from -5% to 15% (mean 6%) with BCAs. BCAs reduce the leakage ratio with robust statistical significance: all parameters being constant in the meta-regression analysis, the ratio drops by 6 percentage points with the implementation of BCAs. In most CGE models, some leakage remains after BCAs implementation, which is not the case with partial equilibrium (PE) models. The most likely explanation is that in CGE models, a part of leakage is due to the international fossil fuel price channel which is unaffected by BCAs, while most PE models do not feature this leakage channel.

Concerning output loss for EITE industries, results are in sharp contrast to results about leakage: CGE models predict loss in a range from 0% to 4% (mean 2%) without BCAs while PE models foresee more than the double. BCAs corrects for the output loss in CGE models but less so in sectoral models. The explanation seems that in PE models, a higher output loss is due to a drop in demand for CO_2-intensive materials, loss which is mitigated by BCAs.

Further, the importance of the coalition size is statistically confirmed and quantified, as well as the impact of extending pricing to all greenhouse gases. The latter reduces the leakage ratio, and the smaller the abating coalition, the bigger the leakage ratio. This meta-analysis also confirms the importance of Armington elasticities in the leakage ratio estimation, a result crucial in terms of uncertainty analysis, which calls for more transparency and sensitivity analyses regarding these parameters in future studies.

The features of BCAs (coverage, level of adjustment, etc.) are of the highest importance for the WTO compatibility, feasibility, and political acceptability. The purpose of the meta-regression was also to assess their impact on competitiveness and leakage. In the meta-regression, the inclusion of all sectors and the presence of export rebates appear to be the two most efficient features to reduce leakage, followed by the adjustment level based on foreign carbon content. Yet one can guess, in the case of hypothetical BCAs implementation, that political and juridical aspects will be the more determinant and that only a "light" version (adjustment based on best available technologies, probably without the inclusion of indirect emissions) is likely to see the light of day.

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2.6 Appendix

2.6.1 Sensitivity analysis

Tables 2.5 and 2.6 present the different sensitivity analyses. Results including or excluding outliers are very similar. The noticeable differences are a slightly higher impact of BCAs for the model including outliers (BCAs diminish the leakage ratio by 7 percentage points instead of 6 percentage points) and less significant coefficient concerning the BCAs features.

Results with the OLS cluster-robust estimation and REML estimation are more diverging. In the OLS cluster-robust model, the impact of extending the coverage to all greenhouse gases is strongly bigger, the impact of the size of the coalition is twice less important and the value of the abatement coefficient is twice more important. Among the BCAs features, the coefficient measuring the effect of covering all sectors instead of only EITE sectors is also twice larger. Remarkably, the impact of BCAs on the leakage reduction is the same for the two models.

2.6.2 Summary statistics

Abating coalition	Europe		Annex 1 without Russia	
Target	15%	30%	15%	30%
No BCA (a)	19%	21%	12%	14%
	[3%;33%]	[3%;38%]	[-5%;28%]	[5%;33%]
BCA light (b)	15%	16%	10%	11%
	[-5%;34%]	[-8%;39%]	[-11%;31%]	[-13%;36%]
BCA strong (c)	9%	10%	4%	6%
	[-16%;33%]	[-18%;38%]	[-22%;30%]	[-24%;35%]

(a) Estimation with the "All" model

(b) Estimation with the "BCAs" model. $AllSect = 1$ only

(c) Estimation with the "BCAs" model. $AllSect = 1$, $Foreign = 1$ and $Exp = 1$

Table 2.4: Benefit transfer: leakage ratio estimations by the meta-regression model

Table 2.5: Sensitivity analysis. Effect of the removal of outliers in the REML estimation

	All		No BCA		BCA	
	Original†	**All Sample**	**Original**†	**All Sample**	**Original**†	**All Sample**
GE	0.091***	0.067	0.047	0.048	0.124***	0.113**
	(2.74)	(1.58)	(1.60)	(1.19)	(4.27)	(2.54)
Coasize	-0.214***	-0.192***	-0.221***	-0.256***	-0.147***	-0.124***
	(12.12)	(8.22)	(10.97)	(10.32)	(5.94)	(4.21)
Abatement	0.090	0.158	0.163*	0.111	0.084	0.203
	(1.04)	(1.36)	(1.78)	(0.93)	(0.69)	(1.41)
Link	0.003	0.004	-0.005	-0.004	0.002	0.013
	(0.26)	(0.33)	(0.48)	(0.29)	(0.13)	(0.78)
GHG	-0.029**	-0.026	-0.014	-0.010	-0.062***	-0.062**
	(2.24)	(1.45)	(1.04)	(0.59)	(2.82)	(2.35)
Armington	0.019***	0.019***	0.033***	0.032***	0.003	0.003
	(4.68)	(3.43)	(7.75)	(5.85)	(0.51)	(0.40)
BCA	-0.063***	-0.074***				
	(14.27)	(12.40)				
Exp					-0.039***	-0.040***
					(2.98)	(2.58)
Foreign					-0.020*	-0.020
					(1.90)	(1.55)
Allsect					-0.042***	-0.030*
					(2.90)	(1.75)
Indirect					-0.015	-0.019
					(0.87)	(0.90)
N	294	310	134	140	160	166
Wald χ^2	386.13	238.64	192.61	147.17	78.25	42.97
LR test	220.50	216.05	96.95	121.06	42.02	62.14

† Some outliers are removed from the sample

Table 2.6: Sensitivity analysis. REML versus OLS cluster-robust estimation

	All		No BCA		BCA	
	REML	**Cluster robust OLS**	**REML**	**Cluster robust OLS**	**REML**	**Cluster robust OLS**
GE	0.091***	0.053	0.047	0.006	0.124***	0.103***
	(2.74)	(1.60)	(1.60)	(0.21)	(4.27)	(3.07)
Coasize	-0.214***	-0.107**	-0.221***	-0.067	-0.147***	-0.105**
	(12.12)	(2.95)	(10.97)	(1.41)	(5.94)	(2.69)
Abatement	0.090	0.197	0.163	0.326**	0.084	0.165
	(1.04)	(0.90)	(1.78)*	(2.22)	(0.69)	(1.12)
Link	0.003	0.016	-0.005	0.020	0.002	0.008
	(0.26)	(0.77)	(0.48)	(1.23)	(0.13)	(0.40)
GHG	-0.029**	-0.083***	-0.014	-0.070***	-0.062***	-0.054***
	(2.24)	(5.01)	(1.04)	(4.89)	(2.82)	(4.51)
Armington	0.019***	0.022***	0.033***	0.034***	0.003	0.006
	(4.68)	(3.36)	(7.75)	(4.36)	(0.51)	(1.60)
BCA	-0.063***	-0.065***				
	(14.27)	(6.63)				
Exp					-0.039***	-0.039*
					(2.98)	(2.03)
Foreign					-0.020*	-0.026**
					(1.90)	(2.31)
Allsect					-0.042***	-0.101***
					(2.90)	(4.01)
Indirect					-0.015	0.001
					(0.87)	(0.08)
N	294	294	134	134	160	160
Wald χ^2	386.13		192.61		78.25	
LR test	220.50		96.95		42.02	
R^2		0.38		0.32		0.59

Table 2.7: Summary statistics for studies. Leakage Ratio

	No BCAs					BCAs				
	Mean	**Range**	**Med**	**Std**	**Obs**	**Mean**	**Range**	**Med**	**Std**	**Obs**
Boh2012	12.2%	(5.0% 23.9%)	12%	5%	15	7.5%	(2.0% 12.0%)	8%	3%	13
Gho2012	10.4%	(2.5% 26.2%)	8%	9%	6	0.8%	(-7.8% 9.5%)	7%	5%	12
AT2012	14.6%	(12.6% 18.0%)	14%	2%	27	10.0%	(9.8% 10.9%)	10%	0%	27
Lan2012	4.4%	(2.0% 9.1%)	4%	2%	20	-6.2%	(-21.0% 4.1%)	-2%	8%	24
Boh2012-2	10.5%	(4.0% 21%)	9%	6%	9	7.9%	(2.3% 19.4%)	5%	7%	9
BalR2012	17.9%	(12% 32.5%)	14%	9%	5	10.8%	(7.0% 19.0%)	9%	5%	5
Wei2012	19.5%	(19.5% 19.5%)	20%	0%	1	17.5%	(17.5% 17.5%)	18%	0%	1
FF2012	13.5%	(7.0% 23.0%)	12%	7%	4	19.0%	(19.0% 19.0%)	19%	0%	1
BB2012	19.5%	(10.0% 32.5%)	18%	9%	3	16.5%	(8.9% 26.9%)	15%	8%	6
Spr2012	14.2%	(13.6% 14.7%)	14%	1%	4	11.4%	(11.1% 11.6%)	11%	0%	3
Car 2012	16.0%	(15.0% 17.0%)	16%	1%	4	12.5%	(11.0% 13.0%)	13%	1%	4
Bed2012	22.4%	(14.3% 38.4%)	19%	9%	12	11.7%	(5.8% 23.0%)	10%	7%	12
Boh2012-3	17.0%	(17.0% 17.0%)	17%	0%	1	12.6%	(8.0% 16.0%)	14%	3%	9
Ant2012	13.5%	(13.5% 13.5%)	14%	0%	1	12.0%	(11.1% 12.9%)	12%	1%	2
Mat2009	4.0%	(4.0% 4.0%)	4%	0%	1	-1.8%	(-7.0% 1.0%)	0%	4%	5
McKW2009	7.5%	(4.0% 11.0%)	8%	5%	2	-22.5%	(-41.0% 22.5%)	-23%	26%	2
PS2007	26.5%	(25.0% 28.0%)	27%	2%	2	24.0%	(21.0% 27.0%)	24%	3%	4
KH2010	11.0%	(11.0% 11.0%)	11%	0%	1	9.0%	(8.0% 10.0%)	9%	1%	2
BabR2005	17.0%	(17.0% 17.0%)	17%	0%	1	16.0%	(16.0% 16.0%)	16%	0%	1
MM2004	24.2%	(0% 41%)	26%	11%	9	-4.5%	(-18.0% 9.0%)	-5%	19%	2
MQ2011-1	8.2%	(7.0% 11%)	8%	5%	10	-0.9%	(-4.2% 3.2%)	-1%	2%	10
DQ2005	13.0%	(13.0% 13.0%)	13%	0%	1	-1.0%	(-6.0% 4.0%)	-1%	7%	2
DQ2008	9.0%	(7.0% 11.0%)	9%	2%	3	-2.0%	(-2.0% 2.0%)	-2%	0%	3
MQ2011-2	10.4%	(10.4% 10.4%)	10%	0%	2	5.5%	(3.8% 7.1%)	2%	2%	4

Table 2.8: Summary statistics for studies. Output change for EITE industries

		No BCAs					BCAs				
	Sector	Mean	Range	Med	Std	Obs	Mean	Range	Med	Std	Obs
Boh2012	”EITE” (1)	-2.86%	(-5.20% -1.20%)	-2.8%	1.1%	13	-1.03%	(-3.00% -0.20%)	-0.9%	0.9%	13
Gho2012	”EITE” (1)	-5.66%	(-7.73% -3.58%)	-5.7%	2.4%	4	0.80%	(0.53%-0.94%)	0.9%	0.2%	4
AT2012	Own Average (7)	-1.83%	(-2.80% -1.00%)	-1.7%	0.9%	3	0.07%	(-0.10% 0.20%)	0.1%	0.2%	3
Lan2012	”EIT sectors” (2)	-2.17%	(-2.70% -1.90%)	-1.9%	0.5%	3	-0.45%	(-0.50% -0.30%)	-0.5%	0.1%	4
Boh2012-2	”EITE” (1)	-2.30%	(-4.95% -0.55%)	-2.2%	1.5%	9	-0.64%	(-1.29% -0.20%)	-0.6%	0.4%	9
Spr2012	”Energy-intensive goods (EIT) (3)	-2.01%	(-2.31% -1.77%)	-2.0%	2.2%	4	-0.27%	(-0.30% -0.23%)	-0.3%	0.0%	3
Car 2012	Own Average (6)	-3.20%	(-3.20% -3.20%)	-3.2%	0.0%	1					0
Mat2009	”Energy-intensive Manufacturing” (4)	-2.30%	(-2.30% -2.30%)	-2.3%	0.0%	5	-0.08%	(-1.50% 2.20%)	-0.3%	1.4%	5
McKW2009	”Non durables” (5)	-0.15%	(-0.20% -0.10%)	-0.2%	0.1%	2	-0.25%	(-0.30% -0.20%)	-0.3%	0.1%	2
PS2007	Own Average (6)	-0.40%	(-0.40% -0.40%)	-0.4%	0.0%	4	0.20%	(0.10%-0.30%)	0.2%	0.1%	4
KH2010	Own Average (8)	-2.00%	(-2.00% -2.00%)	0%	0.0%	2	-0.05%	(-0.60% 0.50%)	-0.1%	0.8%	2
BabR2005	”Energy Intensive Goods” (3)	-1.20%	(-1.20% -1.20%)	-1.2%	0.0%	1	-0.30%	(-0.30% -0.30%)	-0.3%	0	1
MM2004	Steel (only sector)	-7.10%	(-7.10% -7.10%)	-7.1%	0.0%	2	-3.05%	(-4.40% -1.70%)	-3.1%	1.9%	2
MQ2011-1	Own Average (9)	-7.86%	(-7.86% -7.86%)	-7.9%	0.0%	1	-6.41%	(-6.90% -5.80%)	-6.4%	0.4%	5
DQ2005	Cement (only sector)	-7.50%	(-7.50% -7.50%)	-7.5%	0.0%	2	-2.50%	(-3.00% -2.00%)	-2.5%	0.7%	2
DQ2008	Own Average (9)	-11.67%	(-19.00% 5.00%)	-11.7%	7.0%	3	-4.33%	(-7.00% -2.00%)	-4.3%	2.5%	3
MQ2011-2	Own Average (9)	-7.47%	(-7.47% -7.47%)	-7.5%	0.0%	1	-5.47%	(-6.40% -5.00%)	-5.2%	0.7%	4

(1) Aggregation of 5 sectors (Refined goods, Chemical products, Non-metallic minerals, Iron and Steel industry, Non-ferrous metals)
(2) Aggregation of 4 sectors: same as (1) except no Refined goods
(3) 1 sector in the model. Probably same as (2)
(4) Specific sectors non specified
(5) One sector
(6) Iron and Steel and Non-Metallic Minerals
(7) Iron and Steel and Other Non-Metallic Minerals
(8) Steel and Mineral Products
(9) Steel and Cement

Table 2.9: Summary statistics for studies. Welfare variation for the abating coalition

		No BCAs					BCAs				
		Mean	**Range**	**Med**	**Std**	**Obs**	**Mean**	**Range**	**Med**	**Std**	**Obs**
Boh2012	GDP	-0.40%	(-0.40% -0.40%)	-0.40%	0.00%	1	-0.40%	(-0.32% -0.32%)	-0.32%	0.00%	1
Gho2012	Welfare	-0.66%	(-1.58% -0.28%)	-0.54%	0.47%	6	-0.66%	(-0.89% -0.21%)	-0.35%	0.23%	12
AT2012	Welfare	-0.35%	(-0.62% -0.21%)	-0.33%	0.13%	27	-0.35%	(-0.50% -0.25%)	-0.28%	0.11%	27
Lan2012	Welfare	-0.17%	(-0.35% -0.01%)	-0.18%	0.09%	23	-0.01%	(-0.12% 0.09%)	-0.02%	0.04%	24
BalR2012	Welfare	-0.43%	(-0.85% 0.02%)	-0.60%	0.41%	5	-0.43%	(-0.2% -0.40%)	0.01%	0.21%	5
Wei2012	Welfare	-0.50%	(-0.50% -0.50%)	-0.50%	0.00%	1	-0.50%	(-0.4% -0.40%)	-0.40%	0.00%	1
BB2012	Welfare	-0.74%	(-0.88% -0.54%)	-0.79%	0.18%	3	-0.74%	(-0.81% -0.50%)	-0.77%	0.15%	6
Spr2012	Welfare	-0.51%	(-0.54% -0.44%)	-0.54%	0.06%	3	-0.51%	(-0.42% -0.36%)	-0.38%	0.03%	3
Boh2012-3	Welfare	-0.69%	(-0.69% -0.69%)	-0.69%	0.00%	1	-0.69%	(-0.66% -0.42%)	-0.56%	0.08%	9
Mat2009	Welfare	-0.60%	(-0.60% -0.60%)	-0.60%	0.00%	1	-0.60%	(-0.50% -0.40%)	-0.50%	0.04%	5
McKW2009	GDP	-0.60%	(-0.60% -0.60%)	-0.60%	0.00%	2	-0.60%	(-0.60% -0.60%)	-0.60%	0.00%	2
Win2011	Welfare	-0.92%	(-0.92% -0.92%)	-0.92%	0.00%	1	-0.92%	(-0.90% -0.79%)	-0.87%	0.04%	4
BabR2005	Welfare	-0.41%	(-0.41% -0.41%%)	-0.41%	0.00%	1	-0.41%	(-0.32% -0.32%)	-0.32%	0.00%	1

3

Carbon leakage and competitiveness of cement and steel industries under the EU ETS: much ado about nothing

In Chapters 1 and 2, we discussed at lenght the existing evidence about the amount of carbon leakage and losses in competitiveness that can be expected from a given climate policy; and showed that it was not conclusive. Among *ex ante* studies, general equilibrium models point to a positive but limited leakage at the aggregate level (typically from 5% to 25%) while for some carbon-intensive sectors like steel or cement, a higher leakage rate is sometimes forecast (Oikonomou et al., 2006; Demailly and Quirion, 2006). However, the few existing *ex post* studies have not revealed so far evidence of carbon leakage.

The present chapter contributes to the *ex post* literature by econometrically assessing the operational leakage[1] over the first two phases of the EU ETS, in the two

[1] A distinction can be made between leakage that occurs in the presence of capacity constraints

most emitting manufacturing industry sectors: cement and steel. The methodology is to econometrically estimate a relationship, obtained via an analytic model, between net imports (imports minus exports) and the carbon price, controlling for other factors that may influence net imports such as economic activity in and outside Europe. Using two different econometric techniques that provide consistent results, we conclude that net imports of cement and steel have been driven by domestic and foreign demand but not by the CO_2 allowance price, falsifying the claim that the ETS has generated leakage, at least in the short run.

The remainder of the chapter is as follows. Section 3.1 provides a review of the literature on empirical studies focusing on environmental regulations and trade. Section 3.2 gives an overview of the industry contexts of the different studied sectors. Section 3.3 explains the methodology (model and data). Section 3.4 details the results discussed in section 3.5.

3.1 Literature review

Whereas carbon pricing is relatively new, environmental regulations on local pollutants have a much longer history. For example the Clean Air Act was implemented in the US during the 1970s, well before climate change was on the agenda. Therefore the first studies empirically assessing the impacts of environmental regulations on trade dealt with local pollution issues and tested the pollution haven hypothesis/effect (Kalt, 1988; Tobey, 1990; Grossman and Krueger, 1993; Jaffe et al., 1995). The migration of dirty industries to countries with lower environmental standards (pollution havens) depends both on the environmental regulatory gap and on trade tariffs. In the pollution haven hypothesis (respectively effect), the first (respectively the second) factor is held constant[2]. The pollution haven hypothesis was a major concern during the negotiations of the North American Free Trade Agreements in the 1990s (Jaffe et al., 1995), but as the decrease

in the short term, termed operational leakage and leakage which occurs in the longer term via the impacts of the EU-ETS on investment policy, termed investment leakage (Climate Strategies, 2013). Babiker (2005) claimed that investment leakage may occur even with restrictions on international capital flows, through regional domestic adjustements in investments and savings to take advantage of the change in terms of trade.

[2] For a more elaborated presentation and discussion of these notions, cf. Kuik et al. (2013).

in trade tariffs has seemed to slow down, the pollution haven effect has become a more relevant concern (and carbon leakage due the EU ETS would be a "carbon haven effect", see Chapter 1).

The prevailing conclusion of the pollution haven literature is that environmental regulations have a small to negligible impact on relocations (Oikonomou et al., 2006). After a first wave of inconclusive works (Eskeland and Harrison, 2003), a second generation of studies have statistically demonstrated significant but small pollution haven effects using panels of data and industry or country fixed effects (Levinson and Taylor, 2008). Many reasons have been invoked to explain why the widely believed fear of environmental relocations was not observed. Some have pointed out that environmental regulations are not a main driver of relocations contrary to economic growth in emerging countries (Smarzynska, 2002), or that pollution abatement represents a small fraction of costs compared to other costs or barriers that still favor production in industrialized countries (Oikonomou et al., 2006) such as tariffs, transport costs, labor productivity, volatility in exchange rates and political risk. Others highlight that heavy industries are very capital-intensive and tend to be located in capital-abundant countries, or that their capital intensity makes them less prone to relocate than "footloose" industries (**?**). Finally, the Porter hypothesis (Porter and Van der Linde, 1995), implying that regulations bring cost-reducing innovations, has also been cited.

The pollution haven literature is mostly related to command-and-control regulations for local pollutants, whereas the EU ETS is a cap-and-trade system for carbon emissions. Some studies evaluate policies which are closer to the EU ETS such as environmental taxation in some European countries. Miltner and Salmons (2009) studied the impact of environmental tax reforms (ETR) on competitiveness indicators for seven European countries and eight sectors and found that, out of 56 cases, the impact of ETR on competitiveness was insignificant in 80% of cases, positive in 4% and negative for only 16% of the cases (Miltner and Salmons 2009). However, energy-intensive sectors benefited from exemptions and lower rates of taxation. Costantini and Mazzanti (2012) used a gravity model to analyse the impact on trade flows of environmental and innovation policies in Europe and revealed a Porter-like mechanism: when the regulatory framework is followed by private innovation, environmental policies seem to foster rather than undermine export dynamics.

The question of carbon leakage was also a relevant issue for the Kyoto protocol. Aichele and Felbermayr (2012) assessed the impact of the Kyoto protocol on CO_2 emissions, CO_2 footprint and CO_2 net imports, using a differences-in-differences approach with the International Criminal Court participation as an instrumental variable for Kyoto ratification. They concluded that the Kyoto protocol has reduced domestic emissions by about 7%, but has not changed the carbon footprint (CO_2 net imports increased by about 14%). Though they do not explicitly formulate it, their results lead to a carbon leakage estimation of about 100%, contrasting with the other empirical studies. However, two caveats are in order. First, China became a member of the WTO in 2002, just when most developed countries were ratifying the protocol. Since most CO_2 net imports are due to trade with China, the rise in net imports may well be due to China World Trade Organization (WTO) membership rather than to the Kyoto protocol. Second, apart from those covered by the EU ETS, countries with a Kyoto target have not adopted significant policies to reduce emissions in the manufacturing industry. Hence, if Kyoto had caused leakage (through the competitiveness channel), it should show up on the CO_2 net imports of countries covered by the EU ETS rather than on CO_2 net imports of countries covered by a Kyoto target. However, when the authors include both EU membership and the existence of a Kyoto target in the regression, they report that EU membership does not increase CO_2 imports.

Some papers use econometric models to empirically investigate the impact of climate policies on heavy industries *ex ante*, using energy prices as a proxy. Gerlagh and Mathys (2011) studied the links between energy abundance and trade in 14 countries in Europe, Asia and America. They found that energy is a major driver for sector location through specialisation, but they do not quantify relocations under uneven carbon policies. Aldy and Pizer (2011) focused on the US but used a richer sectoral disaggregation. The authors concluded that a \$15 price of CO_2 would not significantly affect the US manufacturing industry as a whole, but that some sectors would be harder hit with a decrease of about 3% in their production.

The EU ETS has constituted a subject of research for a body of empirical studies on different topics: abatement estimation (Ellerman and Buchner, 2008; Delarue et al., 2008), impact of investment and innovation (Calel and Dechezleprêtre, 2014; Martin et al., 2014), distributional effects (Sijm et al., 2006; De Bruyn et al., 2010; Alexeeva-Talebi, 2011), determinants of the CO_2 price (Alberola et al., 2008; Mansanet-Bataller et al., 2011; Hintermann, 2010), but only a limited number of ex post studies have investigated carbon leakage in relation to the EU ETS.

So far, these studies have not revealed any statistical evidence of carbon leakage and losses in competitiveness for heavy industries in the EU ETS. A first strand of literature is based on firm level data. Zachman et al. (2011), using a matching procedure between regulated and unregulated firms, found no evidence that the ETS affected companies' profits; a result that was confirmed by additional studies: Petrick and Wagner (2014) with German firms and Chan et al. (2013) with cement and steel firms. (Dechezleprêtre et al., 2014), using data from the Carbon Disclosure Project, found no evidence that the EU ETS has induced a displacement of carbon emissions from Europe towards the rest of the world. A second strand of literature is based on international trade. Studying the impact of carbon price on trade flows, several studies found no evidence of competitiveness-driven operational leakage for the different sectors at risk of the EU ETS: aluminium (Reinaud, 2008; Sartor, 2013; Ellerman et al., 2010; Quirion, 2011), oil refining (Lacombe, 2008), cement and steel (Ellerman et al., 2010; Quirion, 2011). These results contrast with *ex ante* studies, generally with CGE models (Böhringer et al., 2012) but also with sectoral partial equilibrium models (Monjon and Quirion 2011) that forecast an aggregated carbon leakage ratio in a range of 5% to 25% (Chapter 2) and even more with studies devoted to the cement and steel sectors, which conclude to a leakage ratio in a range of 20% to 60% (BCG, 2008; Oikonomou et al., 2006; Demailly and Quirion, 2006). In most ex ante studies, leakage is partly due to operational leakage and partly to investment leakage. However some studies only assess operational leakage and yet find a significant leakage, including the study devoted to the cement sector by Demailly and Quirion (2006), who find a leakage ratio of around 50%.

Our work goes beyond the above-mentioned studies on several points. First, more data is available as the EU ETS has entered its third phase after eight years of functioning. Second, we introduce a new variable as a proxy for demand outside the EU, which improves the explanatory power of the econometric model. Third, the estimated equations are based on a structural economic model. Finally, we use several time-series regression techniques, which improves the robustness of the results.

3.2 Industry contexts

Cement and steel are both heavy industries affected by the EU ETS. They are the two largest CO_2 emitters among European manufacturing sectors, representing 10% and 9% respectively of the allowance allocations in the EU ETS (Trotignon, 2012). However they rank differently along the two dimensions generally retained for assessing whether a sector is at risk of carbon leakage, i.e. carbon intensity and openness to international trade (Hourcade et al., 2007; Juergens et al., 2013). Cement is very carbon-intensive but only moderately open to international trade while steel features lower carbon intensity but higher trade openness.

3.2.1 Cement

Calcination of limestone and burning of fossil fuel (mainly coal and petroleum coke) to heat material at high temperature make the cement manufacturing process very carbon-intensive (around 0.65 tonnes of CO_2 per tonne[3]). Cement production embodies 5% of worldwide emissions (IEA, 2009).

The raw material of cement, limestone, is present in abundant quantities all over the world. Moreover, the value per tonne of cement is relatively low. Because of these two features, cement is produced in virtually all countries around the world and is only moderately traded internationally (only 3.8% of cement was traded internationally in 2011 (ICR, 2012)). China represents the lion's share of cement consumption and production around the world, due to the large scale developments and infrastructure build-up projects that the Chinese government is undertaking. In 2011, 57% of the 3.6 billion tonnes of cement were produced in China, and the second country producer, India, was far behind with 6% of world production (ICR, 2012).

[3]source: WBCSD, Getting the Numbers Right database, accessed December 2013 (`http://wbcsdcement.org/index.php/key-issues/climate-protection/gnr-database`).

Cement is a sector where international competition is low (Selim and Salem, 2010). Because of low value per tonne and market concentration, important price differences remain even within Europe (Ponssard and Walker, 2008). Prices are higher and producers have more market power inland than near the coasts because transportation costs are much lower by sea than by road.

3.2.2 Steel

Steel is produced either from iron ore and coal using the Blast Furnace - Basic Oxygen Furnaces (BOF) route, for around 70% of world production, or from steel scrap in Electric Arc Furnaces (EAF), for 29% of world production in 2011 (WSA, 2012). The BOF process is roughly five times more carbon-intensive than the EAF but the share of the latter is limited by scrap availability. Steel is very carbon-intensive and accounts for 6% of worldwide emissions (Carbon Trust, 2011b).

Like cement but to a lesser extent, China embodies most of the world steel consumption and production: 45% of the 1,518 million tonnes of 2011 world production (followed by the EU with 12%). Steel has a much higher value-added per tonne than cement (roughly ten times more) and is thus more widely traded. In 2011, 31% of finished steel products were internationally traded (WSA, 2012).

Steel prices, are more homogenous than cement prices and steel futures are sold even on the London Metal Stock exchange. International competition is higher in steel than in cement (Ecorys, 2008).

3.3 Methodology and data

Our goal is to study the impact of carbon price on *competitiveness-driven operational leakage*, at a geographically aggregated level (European Union versus the rest of the world) for two sectors "deemed to be exposed at risk of carbon leakage": cement and steel. If carbon leakage occurs, it is through the trade of these carbon intensive products. An indicator of carbon leakage is then a change in international trade flows (measured by net imports, i.e. imports minus exports).

The short term dynamics of typical cement and steel plant opens the possibility of operational leakage. Indeed, although fixed cost are important, they only represent a minority of total cost: around 11% in the cement sector (BCG (2008) p.

Table 3.1: Summary characteristics of the two sectors. Sources: (WCA, 2011; WSA, 2012; ICR, 2012; CWR, 2011; Carbon Trust, 2011a; Holmes et al., 2011)

	Cement	**Steel**
% of World GHG emissions	5%	6%
Carbon intensity[1]	0.6-0.8	2-4 (BOF)
		0.2-0.9 (EAF)
World Production[2]	3,600	1,514
Top Producer (2011)	China (57%)	China (47%)
Other Main Producers	EU27, India	EU27, Japan, US
Bulkiness ($/tonne)	45-150	500-800
Trade intensity[3]	3.8% (2011)	31.4% (2011)
Market concentration[4]	25%	27.5%
Largest Company	Lafarge	ArcelorMittal
(2011 - % production)	5%	6.4%
International competition	Low	Moderate

[1] Tonnes of CO_2 per tonne of output
[2] 2011. Million tonnes
[3] % of world production traded internationally
[4] Top 10 companies' share in production

13), 21% for the basic oxygen furnace steel production route and 7% for the electric arc furnace steel production route (Schumacher and Sands (2007) p. 813). This allows many plants to be operated well below their nominal capacity (in practice, plants can be operated only during a part of the year and mothballed for the rest of the year). In 2011, 37% of installations in the cement sector and 28% of blast furnaces in the steel sector had emissions between 10% and 75% of their historical emissions level[4], indicating that they were still in operation, but well below their nominal capacity.

3.3.1 Analytical model

We build the simplest possible model capable of featuring carbon leakage. Industries of two regions, e (Europe) and r (rest of the world) are in perfect competition. Therefore the price in each region is equal to the marginal cost. This perfect competition may seem a bold hypothesis, especially for the cement sector, which, in at least some countries, is rather concentrated. However, introducing imperfect competition would significantly complicate the model without necessarily bringing new insights. For example, Cournot competition may reduce the sensitivity of net imports to a price asymmetry and thus leakage, but the results would then become very sensitive to the shape of the demand curve (Demailly and Quirion, 2008).

There is no product differentiation. This assumption, like perfect competition, is chosen for the sake of simplicity. Moreover, we neglect transportation costs for two reasons. First, their introduction would hinder the ability to produce a simple equation to estimate. Second, the estimation of the model with transport costs causes problems of endogeneity (net imports of cement and steel are drivers of shipping costs). Finally, we assume fixed demand, i.e. world demand is not dependent on world price p.

We assume that in region e (Europe), the firm chooses the production level q_e

[4] Capacity utilization rates are not publicly available, but the European Union Transaction Log (`http://ec.europa.eu/environment/ets/napMgt.do`) provides, for each installation covered by the EU ETS, annual emissions and a "historical emissions level" (the median of emissions over 2005-2008 or over 2009-2010, whichever is the highest). Since short-term abatement is limited for a given plant, these data can be used to proxy the evolution of production at the installation level, hence of the capacity utilization rate.

to maximise its profit i.e. its turnover minus production costs (which we assume linear-quadratic with ci_e the intercept and cs_e the slope) plus the value of the free emission allowances, allocated in a quantity a_e which the form cannot influence.[5] The linear part of the cost function includes an extra cost due to the climate policy (CO_{2Cost}), assumed strictly proportional to production. CO_{2Cost} includes the carbon cost times specific emissions plus the abatement cost (if any) per unit produced.

$$\max_{q_e} \pi_e = pq_e - (ci_e q_e + \frac{cs_e}{2} q_e^2 + CO_{2Cost} q_e) + CO_{2Cost} a_e \tag{3.1}$$

Taking the first-order condition and solving for p brings the inverse supply function in Europe:

$$p = ci_e + CO_{2Cost} + cs_e q_e \tag{3.2}$$

The free allocation a_e does not appear in the supply function. Since firms maximise their profit and since allowances are distributed independently of their current decisions, their production level is unaffected by the quantity of allowances they receive for free, if any. More free allowances entail more profit (equation (3.1)), but not more production (equation (3.2)). Yet, a higher CO_2 price yields to a lower production, for a given output price level.

In the rest of the world (region r), firms follow the same program without CO_2 price or allowances, and with different cost parameters.

$$\max_{q_r} \pi_r = pq_r - (ci_r q_r + \frac{cs_r}{2} q_r^2) \tag{3.3}$$

and the inverse supply function is then

$$p = ci_r + cs_r q_r \tag{3.4}$$

in the rest of the world, where q_e and q_r are the *productions* in regions e and r, CO_{2Cost} the carbon cost times specific emissions plus the abatement cost per unit

[5] Note that if allocation were proportional to current production (output-based allocation, the pros and cons of which are discussed in Quirion (2009)), it would enter the supply function, but this is not the allocation method retained in the EU ETS.

produced, if any, ci_e and cs_e (respectively ci_r and cs_r) parameters of the production cost in Europe (respectively the rest of the world).

Trade occurs between the two regions, and we note q_m the net imports from region r to region e. The *demands* in regions e and r are :

$$d_e = q_e + q_m \tag{3.5}$$

$$d_r = q_r - q_m \tag{3.6}$$

$cs_e \times (3.5)$ - $cs_r \times (3.6)$ leads to:

$$(cs_e + cs_r)q_m = cs_e d_e - cs_r d_r - cs_e q_e + cs_r q_r \tag{3.7}$$

Using (3.2) and (3.4) to substitute $cs_e q_e$ and $cs_r q_r$ respectively, and dividing by $(cs_e + cs_r)$ we finally obtain:

$$q_m = \frac{ci_e - ci_r}{cs_e + cs_r} + \frac{1}{cs_e + cs_r} CO_{2Cost} + \frac{cs_e}{cs_e + cs_r} d_e + \frac{-cs_r}{cs_e + cs_r} d_r \tag{3.8}$$

3.3.2 Estimated equation

A reformulation of (3.8) is:

$$\begin{aligned} NImp_{Cement,t} = \quad & \alpha_C CO_2 price_{t-3} + \beta_C Cons_{EU,t-3} \\ & + \gamma_C Ind_{BRICS,t-3} + const_C + \varepsilon_{C,t} \end{aligned} \tag{3.9}$$

for cement, and

$$\begin{aligned} NImp_{Steel,t} = \quad & \alpha_S CO_2 price_{t-3} + \beta_S Ind_{EU,t-3} \\ & + \gamma_S Ind_{BRICS,t-3} + const_S + \varepsilon_{S,t} \end{aligned} \tag{3.10}$$

for steel, where α_C, β_C, γ_C, α_S, β_S, and γ_S are the coefficients to be estimated while $\varepsilon_{C,t}$ and $\varepsilon_{S,t}$ are the residuals, which we assume to be IID, that is later to be tested. The variables are (the source of the data will be detailed in section 3.3.4):

- *Net imports* (*NImp*), or imports minus exports, for each of the two sectors, between the EU27 and the rest of the World. This is the predicated variable, and a proxy for operational leakage. The choice of the geographical delimitation (EU27) is not trivial. Indeed in 2007, the two new member states, Bulgaria and Romania, joined the EU ETS. One year later, the EU ETS welcomed Norway, Iceland and Liechtenstein, countries not in the European Union. As the purpose of this chapter is to study the impact of the EU ETS on competitiveness and leakage, another option was to consider an EU ETS geographical coverage changing over time. This would have posed econometric problems since it would have introduced shocks in the time series. Since these five countries do not produce a significant share of European production, we judge that it was a preferable option not to take these changes into account.

- *CO_2 price*. This is the main regression variable. In the presence of operational leakage due to losses in competitiveness, a positive relationship is expected. Indeed, a high carbon price would induce an increase in the production cost of European products, a loss of market share of European industries vis-à-vis their foreign competitors, and an increase in net imports. We consider the EUA future prices (more details on the series is given in section 3.3.4) rather than spot prices because due to the banking restrictions implemented between 2007 and 2008 (Alberola and Chevallier, 2009), spot prices show a non-reliable behavior during Phase I. Creti et al. (2012); Bredin and Muckley (2011); Conrad et al. (2012) use the future price for the same reason.

- *EU industrial output*, *EU construction index* and *BRICS industrial output* (Ind_{EU}, $Cons_{EU}$ and Ind_{BRICS}). The industrial output is a proxy for the industrial economic activity and therefore the demand side (either domestic or foreign). For cement, we used the European construction index instead of the European industrial output to proxy the demand as construction is the main outlet for cement. We did not find a satisfactory construction index for the BRICS so we used the industrial output for both steel and cement. An increase in local demand is expected to increase the demand of imports and reduce production capacities available for exports. We therefore expect a positive (respectively negative) relation for the European (respectively BRICS) industrial output. We chose to focus on BRICS countries instead of taking an aggregated industrial production index for the rest of the world because to our knowledge such a global index does not exist. Moreover, BRICS countries are the engine of global economic growth: from 8% in 1999, they represented in 2011 20% or the world's GDP. The consumption of cement and steel in BRICS countries (and especially in China) has soared over the last decade. They are not the major destination of EU27 steel exports; however, they are the origin of a noticeable part of EU27 cement and steel imports (China and Russia for steel, China for cement especially between 2005 and 2008) as well as cement exports recently (Russia and Brazil).

To take into account the fact that the potential effect of carbon price on net imports is not instantaneous but necessitates some time (time between production and sale), we introduce a lag in the dependent variables. We select a lag of three months since it brings the best fit,[6] as measured by the usual indicators.[7]

3.3.3 Econometrical techniques

Two aspects are potential barriers to the validity of econometric estimations: endogeneity and the issue of autocorrelation of residuals, since we work on time series data.

First let us consider the thorny issue of endogeneity. It is necessary that variables aimed at explaining the variations of net imports be truly exogenous to validate our econometrical modeling. Such would not be the case if the net imports of cement or steel impacted these explanatory variables. Cement and steel sectors each stand for less than 10% of the covered emissions in the EU ETS. As most of the production is consumed within the EU, variations in net imports induce much less important production variations. It is therefore highly likely that variations in net imports do not affect the carbon price.

[6] The results are robust to a change in the lag (from 1 to 5 months), except for cement in the ARIMA regressions. These results are available upon request.

[7] R^2 for the Prais-Winsten regression and the AIC for the ARIMA regression.

Another source of endogeneity would be that an omitted variable would impact both our main regression variable, the CO_2 price, and the predicated variable. Among the price determinants of the carbon price one can cite the economic activity (which is in the regression with Ind_{EU} or $Cons_{EU}$), political decisions, energy prices (mainly coal and gas[8]) and unexpected weather variations[9] (Alberola et al., 2008; Hintermann, 2010). It seems unlikely that political decisions related to the EU ETS and unexpected weather variations would impact net imports of cement and steel otherwise than potentially through carbon price. Energy prices affect production costs but we suppose that the effect is the same for production outside Europe because prices are determined on a global scale for coal and petcoke, the main energy carriers used for cement and steel production. Therefore, the effect would be compensated between imports and exports.

A simple linear regression would give spurious results because of a strong autocorrelation of error terms, as in many time series data. As the Augmented Dickey-Fuller test shows, all time series are I(1), as we cannot reject the hypothesis of a unit root for the time series but we can for their first difference (see results in appendix 3.6). To treat the question of autocorrelation of residuals, we use two different methods. The first one is the Prais-Winsten estimation, which is an improvement of the Cochrane-Orchutt algorithm.[10] The second one is the classically used model in time series analysis, the ARIMA(p,1,q) model. We identified the ARIMA(p,1,q) process that suits each dependent variable by following the Box and Jenkins methodology and found ARIMA(5,1,3) and ARIMA(6,1,4) for cement and steel respectively. We used the Ljung-Box-Pierce test (which will be explained in further detail in part 3.4.2) to evaluate the results.

[8]An increase in coal price (resp. gas price) makes this source of energy less attractive for electricity production. Therefore the emissions are lower (resp. higher) than expected and the carbon price decreases (resp. increases).

[9]Because unexpected cold waves and heat waves generally induce the use of very carbon intensive power plants.

[10]In a series of recent articles, McCallum (2010), Kolev (2011) and Zhang (2013) have concluded that most so-called "spurious regression" problems are solved by applying the traditional methods of autocorrelation correction, like the iterated Cochrane-Orchutt procedure (Cochrane and Orcutt, 1949). However, other authors, including Martínez-Rivera and Ventosa-Santaulària (2012), Sollis (2011) and Ventosa-Santaularia (2012), have argued that these procedures do not always avoid spurious regressions and propose to pre-test the data and first-differentiate them if they appear to be I(1). Hence we apply both methods in this chapter.

Longer time series give more robust estimations, but including the carbon price for the period 1999-2012 would give spurious results, since there is a break in the time series (this variable is at zero during 1999-2005, then positive). We performed the first regression to have a most robust estimation of net imports depending on local and foreign demand. Then we undertook a second regression for the period 2005-2012 including the carbon price. Comparing the results allows assessing whether the previous estimation is robust in time and examining the effect of adding a carbon price.

3.3.4 Data

All the data are *monthly* from January 1999 to December 2012 (168 points), except for the carbon future prices taken from April 2005 to December 2012 (93 points).

- *Net imports of cement and steel of EU27*. Eurostat international trade database[11]. For cement we take clinker into account, as this semi-finished product is more prone to carbon leakage. For steel, we consider iron and steel in the broad sense, which includes pig iron and semi-finished steel products. The original values in 100kg are converted into Mt/year (with the formula 1Mt/year=833333.3 100kg/month).

- *CO_2 price*. The ICE database[12] provides daily carbon prices beginning on 22 April 2005. To provide a robust price signal, without the effect of banking restrictions for the year 2007, carbon price data of phase I is excluded and the December 2008 futures contract is used throughout the phase I of the scheme until December 14th, 2008. Then, for each day between December 15th of year X-1 and December 14th of year X, we take futures contracts of maturity December 15th of year X. Daily values are then averaged to obtain monthly values from April 2005 to December 2012.

[11] EU27 Trade Since 1988 by HS2, 4, 6 and CN8 dataset (extracted in April 2013). The respective codes for cement and steel are 2523 (cement, including clinker, whether or not coloured) and 72 (iron and steel).

[12] `http://data.theice.com`

- *EU industrial output* and *EU construction index.* Eurostat database.[13] They are both normalized at 2005=100.

- *BRICS industrial output.* Several steps were necessary to compute this index. First, for Brazil, Russia, India and South Africa, the productions in total manufacturing (normalized at 2005=100), were available on the Federal Reserve of Saint Louis Economic Research website[14] derived from the OECD Main Economic Indicators database. China's published industrial statistics are far from being open and the scattered available data are confusing (with changes in variables, in coverage, in measurement, and in presentation). Holtz (2013) reviewed the available official data and constructed a monthly industrial output series. Monthly industrial output (economy-wide constant price) was taken from this paper for the years 1999 to 2011, and extended for the year 2012 thanks to online data[15] giving an annual increase rate of industrial output every month. The obtained data are cyclical: the industrial production is at its highest in December and at its lowest in January and February. We regress the log of this industrial output over time and monthly dummies to estimate seasonal factors, then we withdraw these factors from the original data to obtain a seasonally adjusted Chinese industrial output (which we normalize at 2005=100). Finally, the BRICS industrial output is the weighted mean of national industrial outputs (the weights are the 2005 GDP[16]).

3.4 Results

3.4.1 Descriptive statistics

Future EUA price (see Figure 3.1) oscillated between 20 and 27 euros from June 2005 to June 2006. Then it fell just below 15 euros in the beginning of year 2007

[13]Production in industry and Production in construction- monthly data (extracted in April 2013).

[14]`http://research.stlouisfed.org/fred2/`

[15]`http://www.tradingeconomics.com/china/industrial-production`

[16]United Nations (`http://unstats.un.org/unsd/snaama/dnltransfer.asp?fID=2`)

Table 3.2: Summary statistics of the regression variables

Variable	Obs	Mean	Std dev	Min	(date)	Max	(date)
$NImp_{Cement}$	168	0.19	7.34	-17.95	Jun 2012	19.57	Mar 2007
$NImp_{Steel}$	168	-0.35	13.07	-26.56	Aug 2012	35.62	Jan 2007
CO_2price	93	16.49	5.47	6.79	May 2012	28.09	Apr 2006
$Cons_{EU}$	168	97.10	5.83	84.16	Feb 2012	110.44	Feb 2008
Ind_{EU}	168	99.86	5.52	89.67	Apr 2009	113.10	Jan 2008
Ind_{BRICS}	168	113.38	35.51	63.18	Feb 1999	178.46	Dec 2012

Obs=Observations Std dev=Standard deviation

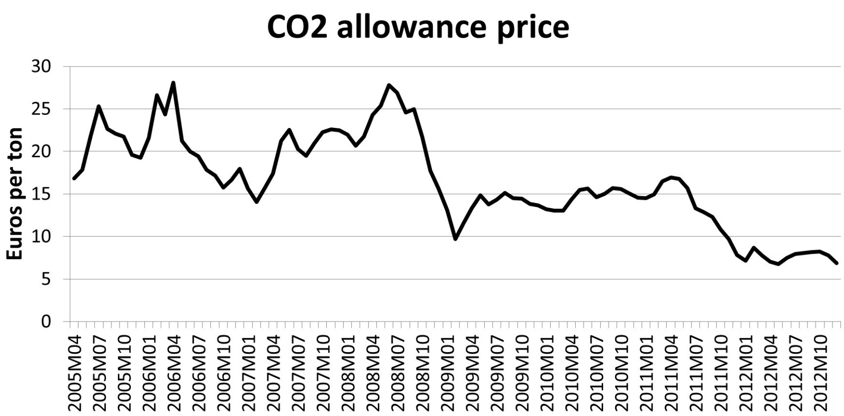


Figure 3.1: CO_2 allowance price used in the regression

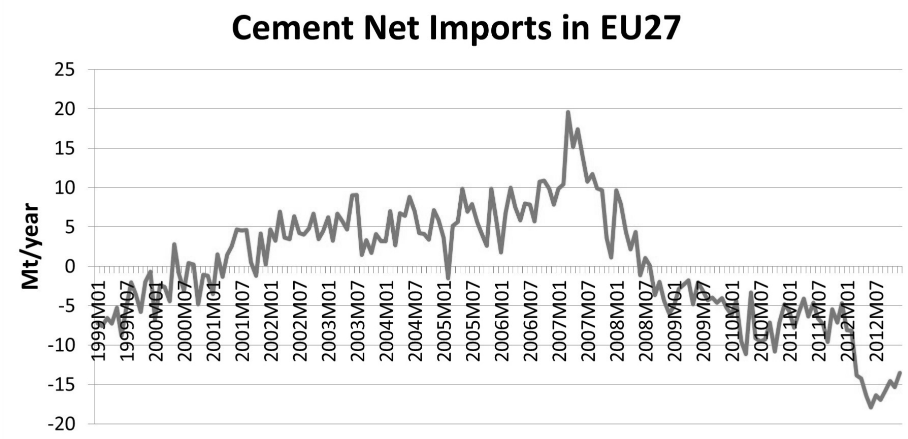
to get back to its original range of values at the end of the same year. In 2008, the EUA future price rose to exceed 25 euros, then fell during the financial crisis down to 10 euros in June 2009. The following two years (until June 2011), the carbon price stabilized between 13 and 16 euros. It fell around 6-7 euros six months later for several reasons (duration of the economic crisis in Europe due to the sovereign debt crisis, interaction with other energy efficiency or renewable climate policies reducing the demand, increasing of allowances), and remained at this price for the year 2012.

The net imports of cement (see Figure 3.2) increasingly rose from 1999 to reach a peak in March 2007 at 20 Mtonnes per year then continuously fell with a recent severe collapse at the beginning of 2012. In 2009, the EU became a net exporter of cement whereas it was a net importer from 2001 to 2007.

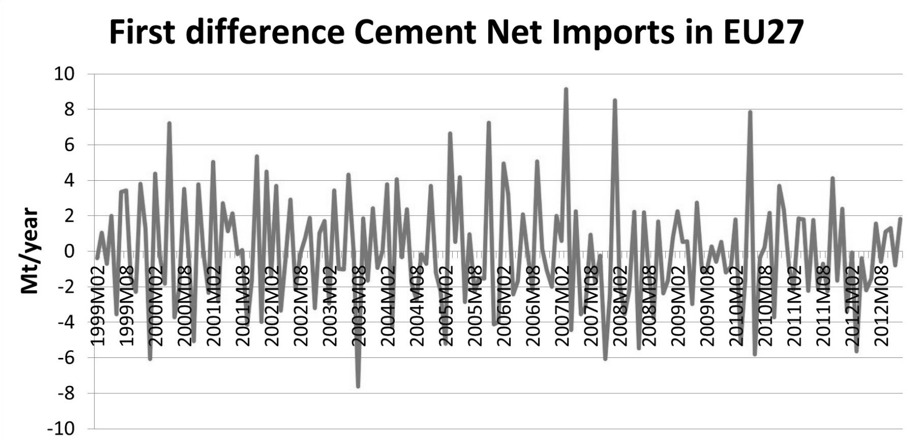
The steel net imports (see Figure 3.3) oscillated at around zero from 1999 to 2005, then the EU became a net importer from 2005 to 2008. Net imports peaked in summer 2007, with 33 Mtonnes per year then collapsed the same year. After a rebound up to 20 Mtonnes per year, the steel net imports fell during the economic crisis. Since then, with the exception of the beginning of 2011, the EU has been a net exporter of steel.

At first sight, cement and steel net imports and the carbon price do not seem highly correlated. The two high carbon price periods (2005-2006 and 2008) most of the time did not coincide with high net imports. On the contrary, for these two products, net imports reached their peak in 2007, while the spot carbon price was very low. Still, it was also a time of intense industrial activity in Europe, a parameter that is taken into account in the regression.

The EU industrial output (see Figure 3.4) increased slightly from 1999 to 2008, then collapsed during the economic recession (by about 20% in six months to go back to its level 10 years before). After a rebound until 2011 without reaching its pre-crisis level, it fell a second time. The EU construction index is very similar, except it plummeted less sharply during the financial crisis, though it never recovered. The BRICS industrial output presents some differences. First, contrary to the EU industrial output, which did not change significantly between 1999 and 2012, the BRICS industrial output almost tripled during the same period. Also hit by the global financial crisis, it took only a year to get back to its pre-crisis

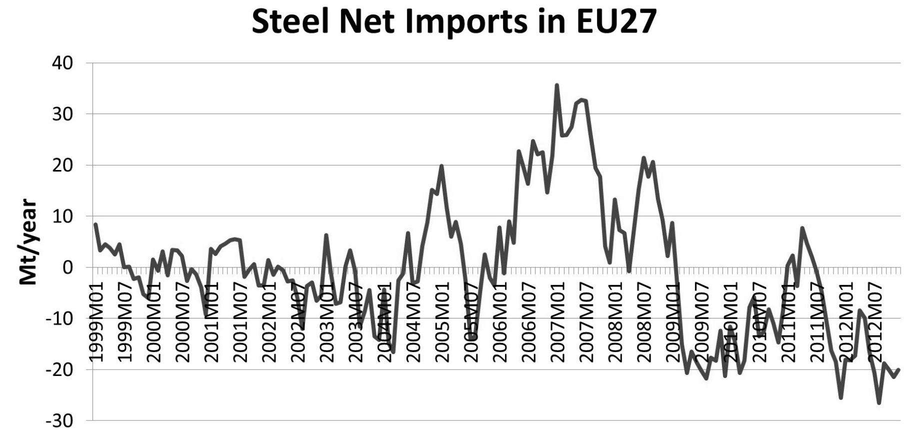


(a) Cement net imports in the EU27

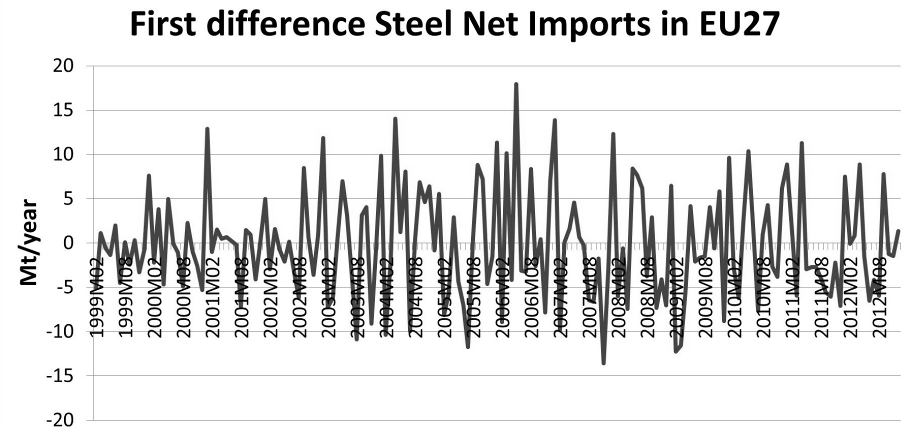


(b) First difference of the cement net imports data series

Figure 3.2: Net Imports (Imports minus Exports) of Cement in the EU27



(a) Steel net imports in the EU27



(b) First difference of the steel net imports data series

Figure 3.3: Net Imports (Imports minus Exports) of Steel in the EU27

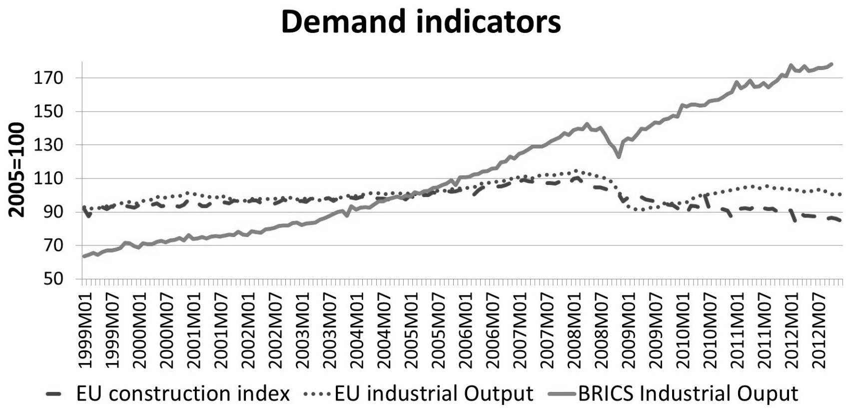


Figure 3.4: Demand indicator variables used in the regression

level. Contrary to its European equivalent, it has grown steadily since then. Year 2011 marked a discrepancy in industrial activities between Europe and the BRICS. Whereas the EU was getting bogged down in a deep economic and industrial recession, the BRICS manufacturing industries were flourishing. For econometric considerations, as the two series before 2011 were much better correlated, this outcome is also of interest to prevent an identification problem.

3.4.2 Regression results

The results are visible in Tables 3.3 and 3.5 for the ARIMA estimations,[17] and in Tables 3.4 and 3.6 for the Prais-Winsten estimations. We recall that for each sector, two estimations are performed, one without the carbon price, for the period 1999-2012, and one with the carbon price for the period 2005-2012 (see section 3.3.3). Comparing the results with the second regression makes it possible to examine the impact of adding a carbon price.

For the Prais-Winsten estimations, we give the value of R^2 and the Durbin-Watson test (which is close to 2 if there is no autocorrelation in the residuals) before and after transformation. In all the Prais-Winsten regressions, the Durbin-Watson tests validate that the residuals are not autocorrelated.

For the ARIMA regressions, the quality of the regressions is assessed with the

[17] For simplicity we do not display the values of the constant term and the ARIMA terms. More detailed results are available upon request.

log-likelihood, the Schwartz and Akaike information criterions (SIC and AIC), and several diagnostic tests for which we report p-values: the Ljung-Box-Pierce statistic (Q test) with a maximum number of lags of 40, the Breusch-Godfrey autocorrelation test, the Engle ARCH test, the Chow break test at time 2005M09 (splitting the full-sample in two sub-samples) and the RESET specification test (against squared estimate alternative).

The null hypothesis in the Ljung-Box-Pierce and Breusch-Godfrey tests is that the observed correlations in the residuals are just a result of randomness. The critical region for rejection of the null hypothesis is when the statistics is higher than the α-quantile of a chi-squared distribution. If we take $\alpha = 5\%$, the model is validated if the p-value is higher than 5%, though a higher p-value indicates that the model is a better fit. All models (cement and steel) appear to be well specified in terms of absence of autocorrelation following this inspection of the residuals' properties. Futher, we accept the null hypothesis of no ARCH effect, the variance is homoskedastic for both the cement and steel models. There is then no further need to investigate the volatility dynamics, which is an intuitive result since we work with monthly data. In addition, the coefficients are stable in all models according to the Chow break test. Finally, for both the cement and steel models, the linear functional form specified in the chapter appears adequate according to the Ramsey RESET specification test.

For cement, in regression (1) in Table 3.3, both $Cons_{EU,t-3}$ and $Ind_{BRICS,t-3}$ are significant, at the 5% and 1% levels respectively. Hence, we verify that indicators of local and foreign demand carry explanatory power in cement net imports. Indeed, an increase in local demand is expected to increase the demand of imports and reduce production capacities available for exports.[18] In our model, an increase of 10 points in local demand[19] would induce an increase of about 3 million tonnes of net imports per year. Moreover, we notice that the signs of the estimated coefficients of $Cons_{EU,t-3}$ and $Ind_{BRICS,t-3}$ conform to the theoretical model (equation (3.8)). The Ljung-Box-Pierce test validates that the residuals of regression (1) are

[18]When carrying Prais-Winsten regressions for imports and exports separately instead of just net imports (results available upon request), we observe that the signs of the demand coefficients still conform to predictions. An increase in local demand induces an increase of imports and a decrease of exports (and the other way around for foreign demand).

[19]The demand was normalized at 100 in 2005.

not autocorrelated.

The coefficients of $Cons_{EU,t-3}$ and $Ind_{BRICS,t-3}$ in regression (2) are very similar to the coefficients of regression (1), and statistically significant at the 5% level. This similarity indicates that the relationship between the cement net imports and local and foreign demand is robust. CO_2price_{t-3} is not statistically significant : the carbon price has no impact on the cement net import variations. The results of (1)bis and (2)bis in Table 3.4 are close to the results of (1) and (2), which is reassuring for the robustness of the results. The coefficients, except for the carbon price, are all significant at the 1% level.

For steel (regression (3) in Table 3.5), $Ind_{EU,t-3}$ is significant at the 1% level, but $Ind_{BRICS,t-3}$ is not statistically significant. For the steel sector, only the local demand carries explanatory power.[20] The impact of the local demand is bigger in the steel industry than in the cement industry[21] compared to the cement net imports: an increase of 10 points in local demand would lead to an increase of about 9 million tonnes in net imports per year. As for cement, the similarity between the results of the two periods implies that the relationship between the steel net imports, local and foreign demand is robust. Similarly to the cement industry, the coefficient of the carbon price CO_2price_{t-3} is not statistically significant.

3.4.3 Robustness and sensitivity analysis

To further assess the robustness of the results presented in Tables 3.3 and 3.5 for cement and steel, respectively, we conduct a recursive analysis. Consider the cumulative sum of $T - \tau$ recursive residuals (Brown et al., 1975):

$$W_t = \sum_{j=\tau+1}^{t} \frac{w_j}{\hat{\sigma}_w} \tag{3.11}$$

[20] Except for the 1999-2012 period for the Prais-Winsten estimation.

[21] The approximate size of cement net imports are a bit more than half as large as steel net imports (see Table 3.2), but the estimated coefficient for local demand (approximated by EU construction or industrial indexes, both around 100) is three times larger.

Table 3.3: Regression estimations. Cement Net Import. ARIMA regressions

Cement	**1999-2012** **(1)**	**2005-2012** **(2)**
$Cons_{EU,t-3}$	0.298**	0.367**
	(0.125)	(0.155)
$Ind_{BRICS,t-3}$	-0.344***	-0.288**
	(0.0998)	(0.125)
CO_2price_{t-3}		-0.0468
		(0.141)
N	164	89
Loglikelihood	-387.06	-209.61
AIC	798.12	445.22
BIC	835.32	477.57
Ljung-Box-Pierce test for white noise residuals		
Ljung-Box-Pierce Test Statistic p-value	0.91	0.86
Breusch-Godfrey autocorrelation test		
LM Test Statistic p-value	0.13	0.22
Engle ARCH test		
LM Test Statistic p-value	0.61	0.10
Chow test for a breakpoint at 2005M09		
Chow Test Statistic p-value		0.33
Ramsey RESET Specification test (against squared estimates alternative)		
Ramsey RESET Test Statistic p-value	0.41	0.62

Table 3.4: Regression estimations. Cement Net Import. Prais-Winsten regressions

Cement	**1999-2012 (1) bis**	**2005-2012 (2) bis**
$Cons_{EU,t-3}$	0.580***	0.432***
	(0.102)	(0.127)
$Ind_{BRICS,t-3}$	-0.100***	-0.240***
	(0.0220)	(0.0499)
CO_2price_{t-3}		-0.0402
		(0.151)
N	165	90
Adjusted R^2	0.25	0.44
R^2	0.26	0.46
ρ	0.71	0.71
DW (original)	0.75	0.67
DW (transformed)	2.34	2.19

where $t = \tau + 1, \ldots, T$ and

$$\hat{\sigma}_w^2 = \frac{1}{T-\tau} \sum_{j=\tau+1}^{T} w_j^2 \tag{3.12}$$

By construction, W_t is a cumulated sum which varies with t. Under the null hypothesis of stability of the coefficients, W_t belongs to the interval $[-L_t, L_t]$:

$$L_t = \frac{a(2t + T - 3\tau)}{\sqrt{T-\tau}} \tag{3.13}$$

with $a = 0.948$ at the 5% level. The null hypothesis of stability of the coefficients estimated is rejected graphically if W_t intersects L_t or $-L_t$. When applying the CUSUM test to the models presented in Tables 3.3 and 3.5, we obtain the plots reproduced in Figure 3.5. Visual inspection confirms the stability of both coefficient estimates, which leads us to validate the specifications chosen for the whole sample of the study. This additional recursive analysis has therefore allowed us to further harness the goodness-of-fit of the specifications retained in Tables 3.3 and 3.5 for cement and steel, respectively.

We then perform some sensitivity analysis on the Prais-Winsten model for both

Table 3.5: Regression estimations. Steel Net Imports. ARIMA regression

Steel	1999-2012 (3)	2005-2012 (4)
$Ind_{EU,t-3}$	1.026***	0.957***
	(0.325)	(0.451)
$Ind_{BRICS,t-3}$	0.118	0.132
	(0.197)	(0.278)
CO_2price_{t-3}		0.444
		(0.319)
N	164	89
Loglikelihood	-501.15	-278.83
AIC	1022.29	577.66
BIC	1053.29	602.55
Ljung-Box-Pierce test for white noise residuals		
Ljung-Box-Pierce Test Statistic p-value	0.10	0.92
Breusch-Godfrey autocorrelation test		
LM Test Statistic p-value	0.18	0.28
Engle ARCH test		
LM Test Statistic p-value	0.11	0.18
Chow test for a breakpoint at 2005-2012		
Chow Test Statistic p-value		0.40
Ramsey RESET Specification test (against squared estimates alternative)		
Ramsey RESET Test Statistic p-value	0.99	0.99

Table 3.6: Regression estimations. Steel Net Import. Prais-Winsten regressions

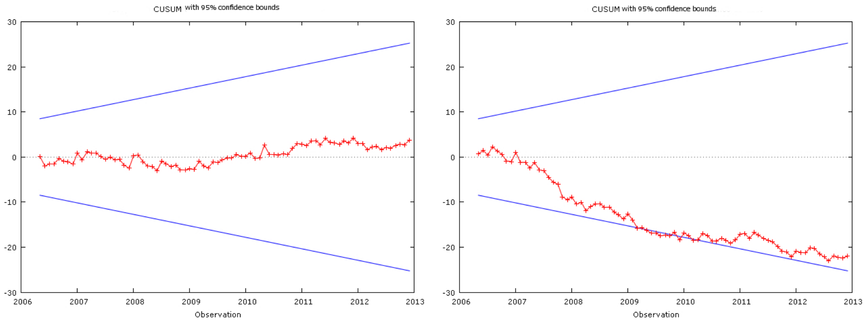
Steel	**1999-2012 (3) bis**	**2005-2012 (4) bis**
$Ind_{EU,t-3}$	1.411***	1.362***
	(0.294)	(0.386)
$Ind_{BRICS,t-3}$	-0.184***	-0.168
	(0.0544)	(0.113)
CO_2price_{t-3}		0.553
		(0.369)
N	165	90
Adjusted R^2	0.13	0.22
R^2	0.14	0.24
ρ	0.76	0.75
DW (original)	0.54	0.53
DW (transformed)	2.09	2.11

cement and steel:

- We select a subcategory of cement and steel products which is more prone to carbon leakage. For cement, we test clinker, the carbon-intensive intermediary product.[22] For steel, we test long products[23] (in opposition to flat products which are higher value-added products) and semi-finished products[24].
- We use an alternative proxy of foreign demand: the computed China industrial output.
- We vary the duration of the original lag by testing two months and four months.

First, a common feature of all these additional regressions is that the carbon price is never statistically significant. The absence of explanatory power of the carbon price on net imports is then very robust across all specifications. Second, variables proxying local and foreign demand remain in most of the cases statistically significant and with the expected signs.

[22] Source Eurostat, HS code 252310.
[23] Source: Eurofer.
[24] Source: Eurostat and Eurofer.



(a) Cement **(b)** Steel

Figure 3.5: CUSUM test for cement and steel

Selecting the subcategories of long and semi-finished steel products gives more statistically significant results (higher R-squared), but it is the opposite for clinker. In addition, semi-finished steel products seem to be more responsive to local demand than long steel products. Further, for both cement and steel, when China (instead of BRICS) industrial output is used to proxy foreign demand, the absolute value of the foreign demand coefficient is lower, while results are slightly more statistically significant. Finally, results remain stable when varying the lag for steel but for cement, results are only stable when switching to two months.

3.5 Discussion

The relationship between net imports and European or foreign demand that was predicted by the analytic model is confirmed by the empirical analysis. An increase in local (respectively foreign) demand increases (respectively decreases) net imports. The fit is particularly good for the cement industry and a little less so for the steel industry. Furthermore, our empirical model does not support the hypothesis that a high carbon price would induce an increase in net imports. For cement and steel, the coefficient of the carbon price has no explanatory power on net imports, even though the CO_2 price has exceeded 20 euros for more than two years during the studied period.

Although based on a longer time series and more elaborate econometric techniques, this empirical work draws the same conclusion as the previous empirical

Table 3.7: Sensitivity Analysis (Cement)

	Cement		Clinker		China Demand		Lag 2 months		Lag 4 months	
	1999-2012	2005-2012	1999-2012	2005-2012	1999-2012	2005-2012	1999-2012	2005-2012	1999-2012	2005-2012
	(1)bis	(2)bis	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)
$Cons_{EU,t-3}$	0.580***	0.432***	0.424***	0.366***	0.559***	0.413***				
	(0.102)	(0.127)	(0.0815)	(0.113)	(0.101)	(0.125)				
$Ind_{BRICS,t-3}$	-0.100***	-0.240***	-0.0735***	-0.184***						
	(0.0220)	(0.0499)	(0.0188)	(0.0451)						
$Ind_{CHINA,t-3}$					-0.0707***	-0.153***				
					(0.0143)	(0.0295)				
CO_2price_{t-3}		-0.0402		-0.0322		-0.0963				
		(0.151)		(0.135)		(0.150)				
$Cons_{EU,t-2}$							0.446***	0.348**		
							(0.111)	(0.134)		
$Ind_{BRICS,t-2}$							-0.0867***	-0.222***		
							(0.0258)	(0.0514)		
CO_2price_{t-2}								0.0358		
								(0.159)		
$Cons_{EU,t-4}$									0.0197	-0.330**
									(0.132)	(0.158)
$Ind_{BRICS,t-4}$									-0.0515	0.175
									(0.0507)	(0.123)
CO_2price_{t-4}										-0.0279
										(0.178)
N	165	90	165	90	165	90	166	91	164	89
Adjusted R^2	0.25	0.44	0.20	0.39	0.27	0.48	0.14	0.39	-0.01	0.02
R^2	0.26	0.46	0.21	0.41	0.28	0.50	0.15	0.41	0.01	0.06
ρ	0.71	0.71	0.75	0.72	0.71	0.69	0.76	0.70	0.89	0.98
DW (original)	0.75	0.67	0.79	0.69	0.73	0.67	0.85	0.83	0.81	0.82
DW (transformed)	2.34	2.19	2.53	2.42	2.33	2.16	2.47	2.38	2.56	2.36

Table 3.8: Sensitivity Analysis (Steel)

	Steel		**Long Steel**		**Semi-Finished Steel**		**China Demand**		**Lag 2 months**		**Lag 4 months**	
	1999-2012	**2005-2012**	**1999-2012**	**2005-2012**	**1999-2012**	**2005-2012**	**1999-2012**	**2005-2012**	**1999-2012**	**2005-2012**	**1999-2012**	**2005-2012**
	(3)bis	**(4)bis**	**(13)**	**(14)**	**(15)**	**(16)**	**(17)**	**(18)**	**(19)**	**(20)**	**(21)**	**(22)**
$Ind_{EU,t-3}$	1.411***	1.362***	0.269***	0.241***	0.854***	0.771***	1.385***	1.304***				
	(0.294)	(0.386)	(0.0695)	(0.0883)	(0.141)	(0.159)	(0.275)	(0.380)				
$Ind_{BRICS,t-3}$	-0.184***	-0.168	-0.0738***	-0.0970***	-0.126***	-0.129***						
	(0.0544)	(0.113)	(0.0129)	(0.0255)	(0.0236)	(0.0476)						
$Ind_{CHINA,t-3}$							-0.128***	-0.107				
							(0.0333)	(0.0667)				
CO_2price_{t-3}		0.553		0.101		0.297		0.560				
		(0.369)		(0.0900)		(0.208)		(0.370)				
$Ind_{EU,t-2}$									1.558***	1.604***		
									(0.281)	(0.368)		
$Ind_{BRICS,t-2}$									-0.192***	-0.254**		
									(0.0512)	(0.106)		
CO_2price_{t-2}										0.218		
										(0.362)		
$Ind_{EU,t-4}$											1.378***	1.350***
											(0.307)	(0.379)
$Ind_{BRICS,t-4}$											-0.202***	-0.264**
											(0.0582)	(0.111)
CO_2price_{t-4}												0.474
												(0.367)
N	165	90	165	90	165	90	165	90	166	91	164	89
Adjusted R^2	0.13	0.22	0.17	0.30	0.22	0.44	0.15	0.23	0.16	0.27	0.12	0.25
R^2	0.14	0.24	0.18	0.32	0.23	0.46	0.16	0.26	0.17	0.29	0.13	0.28
ρ	0.76	0.75	0.76	0.71	0.61	0.49	0.75	0.75	0.75	0.73	0.77	0.74
DW (original)	0.54	0.53	0.55	0.76	0.79	1.01	0.56	0.55	0.53	0.54	0.52	0.52
DW (transformed)	2.09	2.11	2.13	1.95	2.07	2.08	2.08	2.11	2.04	2.05	2.06	2.04

literature (Reinaud, 2008; Lacombe, 2008; Sartor, 2013; Ellerman et al., 2010; Quirion, 2011), which is that the EU ETS has not induced operational carbon leakage.

Some may argue that, because these industries have benefitted from a large over-allocation of allowances during this period, the risk of operational carbon leakage was null. Yet, as long as the allowances are allocated independently of current output, the operator of an installation may reduce emissions (by increasing the CO_2 efficiency of its production process or by reducing the output level) in order to sell allowances even though he has received more allowances than its emissions (Montgomery, 1972; Quirion, 2009). This is why, in the simple model presented in section 3.3.1 and in richer models as well (e.g. Goulder et al. (2010)), the amount of allowances freely allocated (if any) impacts the profit level but not the output level. Numerous studies in experimental economics validated the theoretical equivalence between auctioning and grandfathering (Goeree et al., 2010; Wråke et al., 2010; Camacho-Cuena et al., 2012; Grimm and Ilieva, 2013). Further, evidence of positive pass-through rate of the carbon price provided by econometric studies (mostly in the power sector (Sijm et al., 2006), but also in some manufacturing sectors (Alexeeva-Talebi, 2011; De Bruyn et al., 2010)) was an indirect proof that companies took into account the full cost of free allowances: when setting their output price, companies included the value of CO_2 emissions in their products' marginal cost, even though they received for free approximately enough allowances to cover their emissions.

Hence, if one considers that companies behave as profit-maximisers, the over-allocation of allowances should not have an influence on operational leakage.[25] The outcome of this study (no operational carbon leakage) is then far from trivial. It involves that in the price range that has been experienced for CO_2 (below 30 euros per ton), operational carbon leakage is not a serious threat for the energy-intensive industries.

This result applies in theory regardless of the allowances balance. In practice, no shortage of allowances is likely to occur until at least 2020: cement and steel

[25] As long as the allocation of free allowances is independent from the current production level (which is the case under the EU-ETS for phases I and II), otherwise it would impact the production level of firms.

companies have banked a significant surplus of allowances,[26] and they will still benefit from over-allocation in phase III unless demand goes back to pre-crisis levels.[27] Thus auctioning a part of the allowances currently freely allocated to these sectors would not entail carbon leakage while it would bring public revenues, which would be welcome especially considering the public debt faced by many European countries.

However, the impact of the EU ETS on investment leakage, which corresponds to changes in production capacities as the result of the EU's climate policy, is still an open question. Indeed, since 2013 (the start of the EU ETS third phase), less allowances are allocated if current production falls through a threshold: 50% (respectively 25% and 0%) of the otherwise planed allocation if production falls below 50% (respectively 25% and 10%) of its historical emissions level. Moreover, allowances are allocated for new production capacities and for capacity extensions (Chapter 5). Thus free allowances might mitigate carbon leakage through an impact on the production capacity in Europe, rather than through operational decisions. This question could be investigated using foreign direct investment data as in the original pollution haven literature. Pending such investigation, we cannot conclude that free allocation should be scrapped, even though carbon leakage is presented as the main argument to maintain them.

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[26]Based on EUTL data at the installation level, we estimate the allowances surplus in the beginning of year 2014 at respectively two and two and a half years worth of pre-crisis emissions for the cement and the steel sector. If we suppose that companies have sold half of this surplus to acquire liquidities, they still have more than one year worth of emissions in banked allowances, corresponding to ten years facing a 10% allowances shortage.

[27]Allocation in phase III is based on a stringent benchmark and a cross sectoral correction factor, but multiplied by a historical activity level (HAL), which corresponds roughly to pre-crisis production. Based on EUTL data, we find that 2013 allocations were 25% higher than emissions in the cement sector and 12% to 35% in the steel sector (whether or not emissions from power plants using waste gases are taken into account).

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3.6 Appendix

Table 3.9: Augmented Dickey-Fuller (ADF) tests for unit root

	1999-2012		**2005-2012**	
	Test Statistic	**p-value**	**Test Statistic**	**p-value**
$NImp_{Cement}$	-2.527	0.1091	-1.441	0.5627
$\triangle NImp_{Cement}$	-18.849	0.0000	-13.185	0.0000
$NImp_{Steel}$	-2.881	0.0477	-1.777	0.3918
$\triangle NImp_{Steel}$	-14.655	0.0000	-10.448	0.0000
Ind_{EU}	-1.709	0.4266	-0.938	0.7752
$\triangle Ind_{EU}$	-11.065	0.0000	-6.793	0.0000
Ind_{BRICS}	0.730	0.9904	-0.578	0.8759
$\triangle Ind_{BRICS}$	-17.279	0.0000	-12.159	0.0000
$Cons_{EU}$	-1.425	0.5702	-0.739	0.8364
$\triangle Cons_{EU}$	-18.429	0.0000	-13.467	0.0000
CO_2price			-1.098	0.7158
$\triangle CO_2price$			-7.966	0.0000

The ADF model specified is model with constant

4

Reaping the carbon rent: Abatement and overallocation profits in the European cement industry, insights from an LMDI decomposition analysis

Cement is the most widely-used man-made material in the world (Moya et al., 2010), and also one of the most carbon-intensive products. The manufacture of cement accounts for approximately 5% of global anthropogenic emissions (IEA, 2009). China has the lion's share of cement production with 58% of the 3,700 million tons produced in 2012. The European Union is now the third-biggest producer with 5% of global production, behind India with 7% (U.S. Geological Survey, 2013).

Since 2005 European cement emissions have been covered by the European Union Emission Trading Scheme (EU ETS), presented as Europe's flagship policy to tackle climate change (Branger et al., 2015). In this cap-and-trade system, installations can buy or sell tradable allowances to attain emissions caps. A key feature of the EU ETS, the question of whether allowances should be auctioned or received free of charge (and in the latter case, what should be the allocation plan, or the number of allowances per installation), has proved to be a very controversial topic (Boemare and Quirion, 2002; Ellerman and Buchner, 2007). While most economists favored auctioning, the European Union opted for almost completely free allocation for all sectors (industry and power sector) during phase I (2005-2007) and phase II (2008-2012); and maintained completely free (but declining at 1.74% per year) allocations[1] in phase III (2013-2020) for sectors "deemed to be exposed to carbon leakage", and partly free for the rest of manufacturing industry (European Commission, 2009).

Indeed, the main argument used to justify free allocation has been the preservation of heavy industries' competitiveness and the prevention of carbon leakage, which is a shift of emissions from carbon-constraint countries to less carbon-constrained countries induced by asymmetric carbon pricing (Dröge, 2009). However, economic theory suggests that free allocation, if independent from current production, is inefficient at preventing leakage in the short term and would only provide a disincentive to plant relocation (Wooders et al., 2009). In other words, in the short run free allocations would compensate firms for profitability losses without addressing market share losses and carbon leakage (Cook, 2011).

In addition to generous allocation caps, the economic downturn after 2008 led to a decrease in industrial production, which generated a large surplus of allowances in the market. These financial assets have mainly been held by cement and steel companies, because electricity demand has been much less impacted by the economic downturn.

[1] Allocations in phase III are based on a product benchmark (the average of the 10% best performing installations: 766kg CO2 per ton of clinker, the CO2-intensive intermediate product required to produce cement), multiplied by a "cross sectoral correction factor" (0.9422 in 2013, declining by 1.74% per year), historical activity level (HAL, a formula leading approximately to pre-crisis level of production), and an "activity level correction factor" (reducing allocations by half or four if the plant is functioning below 50% or 25% of its HAL). Completely free allocations are then maintained though the overall cap of allowances is less generous (it has been reduced by 23% between 2012 and 2013) and declining. However because actual production is much lower than pre-crisis level, 2013 emissions were 20% lower than the cap.

Instead of suffering from financial losses, energy-intensive industries seem to have thrived from the scheme. Sandbag, a non-governmental organization, has estimated that the ten "carbon fat cats" have reaped billions of Euros in windfall profits (Sandbag, 2010). However, their analysis, based on the European Union Transaction Log (EUTL) data, is based upon equivalence between allowances surplus and overallocation, without considering the fact that some allowances are obtained by reducing the carbon content of industrial products (Ellerman and Buchner, 2008). Indeed, apart from financial outcomes, an important question remains: whether the EU ETS has fulfilled its original purpose which was to trigger a transition towards low-carbon industry.

Studies assessing abatement in the manufacturing industry have obtained mixed results (Neuhoff et al., 2014). Zachman et al. (2011) find a significant reduction in carbon intensity for basic metals (whose emissions occur mostly in the steel sector) and non-metallic minerals (whose emissions occur mostly in the cement sector) between 2007 and 2008 compared to 2005-2006. Yet (Kettner et al., 2014) find very limited reduction in carbon intensity in the cement and lime sector, and attribute most of it to an increase in clinker imports – which implies carbon leakage. Moreover Egenhofer et al. (2011) find almost no decrease in the manufacturing industry's carbon intensity in 2008, which seems to contradict Zachman et al. (2011) results.

In this chapter, we propose to shed light on the questions of abatement and overallocation in the European cement industry, exploiting EUTL data, Eurostat international trade data, and the detailed and comprehensive Getting the Numbers Right (GNR) database from the Cement Sustainability Initiative (CSI). We perform an LMDI (Log Mean Divisia Index) decomposition (Ang, 2004) of emissions due to cement production in Europe. We measure the impact of seven effects on emissions variations, which correspond to different mitigation levers: activity, clinker trade, clinker share, alternative fuel use, thermal and electrical energy efficiency, and decarbonisation of electricity. This analysis allows us to identify the key drivers behind changes in aggregated carbon emissions, in the EU 27 as a whole and in six major European producers: Germany, France, Spain, the UK, Italy and Poland.

A distinction can be made between the first two effects (activity and clinker trade) that generate non-technological abatement and the others that generate technological abatement. Making assumptions on counterfactual scenarios, we estimate the technological abatement induced by the EU ETS and break down its main factors. Furthermore, our emissions decomposition model allows us to identify which part of the allowances surplus (allocations minus emissions) is due to technological performance and which is due to a change in activity or clinker outsourcing. We are then able to compute overallocation and "overallocation profits".

We find that the EU ETS has induced a small but positive abatement of 26 Mtons of CO_2 ($\pm$16 Mtons) from 2005 to 2012 (corresponding to a 2.2% $\pm$ 1.3% decrease), mostly thanks to the reduction in the clinker-to-cement ratio. However we cannot rule out another explanation, i.e. the massive increase in steam coal and petcoke prices in the 2000s (**?**). This aggregate figure hides important differences at national levels. Whereas technological abatement has been important in Germany (5% $\pm$ 3%) and in the UK (4% $\pm$ 3%), it has been small in France, and insignificant or negative in Spain, Italy and Poland. In addition, we estimate that the European cement industry has reaped 3.5 billion Euros of overallocation profits during phases I and II. Most of these profits come from the economic downturn that has reduced the demand for cement and thus for cement production, in turn generating a massive surplus of allowances.

The rest of the chapter is structured as follows. Section 4.1 details the cement manufacturing process and the mitigation options. Section 4.2 explains the emissions decomposition methodology. Section 4.3 applies this decomposition to changes in emissions in the European cement industry from 1990 to 2012. Section 4.4 is an assessment of technological abatement induced by the EU ETS and of overallocation profits. Section 4.5 concludes.

4.1 Mitigation options in the cement industry

4.1.1 Cement manufacture at a glance

Cement manufacture can be divided into two main steps: clinker manufacture, and blending and grinding clinker with other material to produce cement.

Clinker is produced by the calcination of limestone in a rotating kiln at 1450 degrees Celsius. Carbon dioxide is emitted in two ways. First, the chemical reaction releases carbon dioxide (ca. 538 kgCO_2 per ton of clinker[2]) which accounts for roughly two thirds of carbon emissions in clinker manufacture. The remaining CO_2 comes from the burning of fossil fuel to heat the kiln. The fuels used are mostly the cheapest ones, petcoke and coal (the use of gas and oil is precluded by cost, except in some locations where they are very cheap, which is not the case in the EU).

Raw material preparation, kiln operation, blending and grinding consume electricity which causes indirect emissions. However, nearly all carbon emissions (around 95%) in cement manufacture come from direct emissions in clinker manufacture.

To reduce emissions from cement production,[3] various options are thus available:

(i) *Reduction of cement production*, which may be due to reduced activity in the construction industry, to leaner structures or to the substitution of alternative materials to cement.

(ii) *Clinker substitution*. Since clinker manufacture is the most carbon intensive part of cement manufacture, partially substituting some other material for clinker is an efficient way to reduce emissions per ton of cement produced. The most common type of cement, ordinary Portland cement, is produced by mixing 95% of clinker and 5% of gypsum, but the clinker-to-cement ratio is lowered in blended cements.

(iii) *Clinker outsourcing*. This is a way to reduce emissions within a given geographical perimeter, but emissions then occur elsewhere, which causes carbon leakage.

(iv) *Alternative fuel use*, which releases less CO_2 for the same calorific value pro-

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[2]The process CO_2 emission factor is generally considered as a fixed factor. However it is slightly variable mainly because of the ratio of calcium carbonate and magnesium carbonate in the limestone. When process emissions are actually measured, a narrow peak in the distribution can be observed at 538 kgCO_2 per ton of clinker (Ecofys et al. (2009) Figure 2). However, the factor used in the EU ETS Monitoring and Reporting of Greenhouse gas emissions (MRG) is only 523 kgCO_2 per ton of clinker, derived from IPCC methodology.

[3]If we consider cement *consumption* and not cement *production*, another option can be added: cement outsourcing. We performed the same analysis for cement consumption with a more complicated decomposition, adding cement trading. As the results barely changed (the cement trading effect represented less than 3 Mtons of CO_2 or 2% of emissions), for the sake of simplicity we only retained the analysis of cement production.

duced.

(v) *Energy efficiency*, which can be divided into two parts, *thermal* energy efficiency and *electrical* energy efficiency.

(vi) *Decarbonisation of the electricity*.

(vii) *Carbon capture and storage*.

(viii) *Innovative cements*, or carbon neutral cements based on totally different processes.

The next section details these options, which do not have the same status. Lever (i) is driven by cement demand and is not a direct choice made by cement companies. Levers (ii) to (v) are operational options used by cement companies (though lever (iii) does not reduce global emissions, it can be a rational choice for a company covered by an emissions trading scheme). Lever (vi) is beyond the scope of cement producers, and depends on electricity producers (which have an incentive to use it when there is a price on carbon). Abatement due to levers (i) to (vi) will be empirically assessed in this study. Levers (vii) and (viii) are in the research and development stage. Though promising, these options have not generated abatement yet.

The challenge of a non-global climate policy is to induce all these options (except (iii)) without generating clinker or cement imports, which would lead to carbon leakage.

4.1.2 Data sources

The work of this chapter is based on the cross-referencing of three databases:

- the *Getting the Numbers Right* (GNR) database[4] (WBCSD, 2009) developed by the Cement Sustainability Initiative (CSI), operating under the World Business Council on Sustainable Development (WBCSD).
- the *European Union Transaction Log*[5] (EUTL) which is the registry of the EU ETS, and provides allocations and verified emissions at the installation level.

[4] `http://wbcsdcement.org/GNR-2011/index.html`. Variables have names we will refer to for data sourcing. For example the clinker-to-cement ratio is variable 3213.

[5] `http://ec.europa.eu/environment/ets/napMgt.do`

- the *Eurostat* international trade database[6] for clinker trading.

The GNR database covered 94% of European cement production in 2012 (only minor producers with small production volumes are excluded), which is remarkably high. Data are available[7] for 1990, 2000, and 2005 to 2012. Data can be obtained at the EU 28 level and at the national level for big producers (so we have used data for Germany, France, Spain, the UK, Italy and Poland). Although the GNR database contains data on production and emissions, we use this database for its intensity (i.e. rate-based) indicators in the cement industry, for reasons related to coverage and methodology (see part 4.2.1). A performance indicator not included in GNR, the electricity emission factor, comes from the Enerdata database.[8]

The cement sector is a subsector of the cement/lime EUTL sector (47% of installations and 90% of allocations). We have collected plant-by-plant information on 276 cement plants with kilns covered by the EU ETS. Some characteristics of our cement EUTL database, which are in line with Table 1.2 in European Commission (2010) and Table 4 in Moya et al. (2010),[9] are given in Table 4.1. The match between EUTL emissions and GNR gross direct emissions is good but not

[6] `http://epp.eurostat.ec.europa.eu/newxtweb/setupdimselection.do`, EU Trade since 1988 by HS2, 4, 6 and CN8 dataset (extracted in February 2014). The code for clinker is 252310 ("cement clinkers").

[7] Due to confidentiality reasons, there is a one year gap between data collection and publication. The latest available data (2012) were published in August 2014.

[8] `http://www.enerdata.net/enerdatauk/knowledge/subscriptions/database/energy-market-data-and-co2-emissions-data.php`

[9] Their list contains 268 installations in 2006 at the EU 27 level (compared to 270 for us (Norway has two cement plants and Croatia four)). There are some discrepancies for France (33 in instead of 30 for us), Germany (38 instead of 43), Italy (59 instead of 52), and some other countries (1 plant difference). In Germany, geolocalization of plants revealed that three plants had two EU ETS installations and 1 had three. Our list was cross-checked with the Cemnet database (`http://www.cemnet.com/GCR/`), Sandbag database, and public reports of major cement companies.

perfect.[10] In addition, we use the Sandbag database[11] for offset credits utilization at the installation level.

Whereas total imports and exports are directly available in the Eurostat international trade database at the EU 27 level, they have to be computed from country-pairs raw data at the national level. Also, some corrections needed to be made to take into account the changing geographical perimeter of the EU ETS. Because they are absent from Eurostat, we used the Comtrade database[12] for net imports in Norway, Iceland, and the EU 27 before 1999.

4.1.3 Clinker substitution

Reducing the clinker-to-cement ratio is a very efficient abatement option since most of the carbon emissions are produced during clinker manufacturing. The most-used clinker-substituting materials are fly ash (a residue from coal-fired power stations), ground blast furnace slag (a by-product of the steel industry), pozzolana (a volcanic ash) and limestone. Blast furnace cement offers the highest potential for clinker reduction with a clinker-to-cement ratio of 5-64%, compared to pozzolanic cement (45-89%) and fly ash cement (65-94%) (Moya et al., 2010).

Two barriers are impeding the deployment of blended cements. The first is the regional availability of the clinker substitutes, or their price (since these products have low value per ton, transportation costs are high). The phasing out of coal-fired plants triggered by climate policy will make fly ash scarcer. Ground blast depends on iron and steel production, and pozzolanas are present only in certain volcanic regions (mainly Italy). Second, the physical properties of these

[10]GNR emissions are higher in the United Kingdom , Germany and Poland, (3% on average respectively for all) whereas they are lower Italy and at the EU 27 level (6% and 2% respectively). France and Spain are perfect matches. Besides data-capture errors, differences in emissions can occur for different reasons. First, there is a mismatch in installations covered. GNR contains more plants because it includes grinding or blending plants, but some plants with kilns are not covered, so emissions at the national level have to be extrapolated. Second, accounting methodologies are different. Process emissions are measured in GNR (there is a peak in the distribution at 538 kgCO_2 per ton of clinker see figure 2 in (Ecofys et al., 2009)) whereas a default factor derived from IPCC methodology of 523 kgCO_2 per ton of clinker is used in the EU ETS. Non-kiln fuels are not reported in some countries for the EU ETS but are (partially) reported in GNR. The carbon content of alternative fuels is also accounted for differently.

[11]`http://www.sandbag.org.uk/data/`

[12]`https://wits.worldbank.org/WITS/WITS/Restricted/Login.aspx`

Table 4.1: Cement EUTL database. Country level (Sandbag database used for offset credits)

	Number of plants		Annual Allocation ($MtonsCO_2/year$)		Annual Emissions ($MtonsCO_2/year$)		Offset credits used (phase II)	
Region	**Phase I**	**Phase II**	**Phase I**	**Phase II**	**Phase I**	**Phase II**	**Total**	**% cap**
EU ETS	254	262	158.3	178.6	153.8	131.1	88.6	10.0%
Germany	47	40	23.5	21.0	20.9	19.6	22.2	21.1%
France	30	30	14.1	15.3	14.3	12.5	10.1	13.3%
Spain	35	36	27.5	29.2	26.8	17.6	10.7	7.4%
United Kingdom	13	15	5.6	10.1	5.7	6.3	3.3	6.7%
Italy	52	54	26.2	28.0	27.8	21.1	10.0	7.2%
Poland	11	11	10.8	11.0	9.7	10.0	4.6	8.4%
Subtotal	189	187	108.2	115.0	105.8	87.5	61.2	10.6%
	74%	71%	68%	64%	69%	66%	69%	
Austria	9	9	2.8	2.7	3.0	2.6	1.2	9.2%
Greece	8	8	11.1	10.8	10.7	6.6	3.3	6.2%
Romania	7	7	2.3	9.3	2.2	5.1	4.3	9.3%
Czech Republic	6	5	3.0	2.8	2.9	2.5	1.4	10.0%
Portugal	6	6	6.8	6.7	6.6	5.1	3.3	10.0%
Belgium	5	5	5.5	5.0	5.0	4.3	1.0	4.0%
Bulgaria	NP	5	NP	3.6	NP	1.9	2.1	11.5%
Hungary	4	5	2.4	2.2	2.1	1.4	0.9	8.4%
Ireland	4	4	3.6	4.0	3.8	2.1	2.2	11.0%
Slovakia	4	4	2.3	2.7	2.2	1.9	0.9	6.7%
Sweden	3	3	2.2	2.5	2.2	2.2	1.2	9.3%
Cyprus	NP	2	NP	1.7	NP	0.9	0.8	9.3%
Finland	2	2	1.1	1.3	1.0	0.8	0.5	8.0%
Netherlands	1	1	0.8	0.7	0.6	0.5	0.3	8.8%
Slovenia	2	2	0.8	0.7	0.8	0.6	0.4	12.1%
Denmark	1	1	2.8	2.6	2.7	1.7	0.8	6.5%
Estonia	1	1	0.9	0.8	0.9	0.8	0.2	4.1%
Latvia	1	2	0.3	0.9	0.3	0.6	0.4	8.9%
Lithuania	1	1	1.1	1.0	1.0	0.7	1.0	20.0%
Luxembourg	1	1	0.8	0.7	0.7	0.6	0.4	10.0%
Norway	NP	2	NP	1.3	NP	1.2	0.8	12.7%

Note: NP is for Non Participating

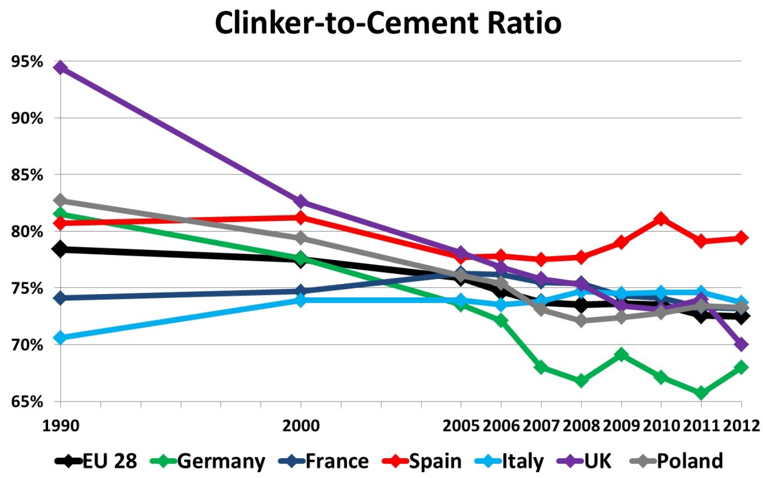

Figure 4.1: Clinker-to-cement ratio for the EU 28 and main European countries. Source: WBCSD GNR Database, variable 3213

alternative cements such as strength, colour and workability, and their acceptance by construction contractors, constitute another barrier to their implementation (IEA, 2009).

Figure 4.1 displays the clinker-to-cement ratio in 1990, 2000 and from 2005 to 2012 for the European Union (with 28 member states) and the six biggest cement producers in Europe: Germany, France, Spain, Italy, the United Kingdom and Poland. The average EU 28 clinker-to-cement ratio decreased from 78% in 1990 to 73% in 2012. The UK is the country for which the clinker-to-cement ratio has decreased the most dramatically, from 95% in 1990 to 70% in 2012. In 2012 Germany was the country with the lowest clinker-to-cement ratio, 68%, whereas Spain had the highest, 79%.

4.1.4 Clinker outsourcing

Clinker outsourcing is a drastic method to reduce carbon emissions within a given geographical perimeter, but it does not in general reduce emissions on a global scale (carbon intensity is approximately the same in Europe and abroad and this adds emissions due to transportation). The increase in emissions abroad due to a regional climate policy is called carbon leakage (Reinaud, 2008). In the EU ETS, free allocation of allowances was presented as a way to mitigate the risk of leakage.

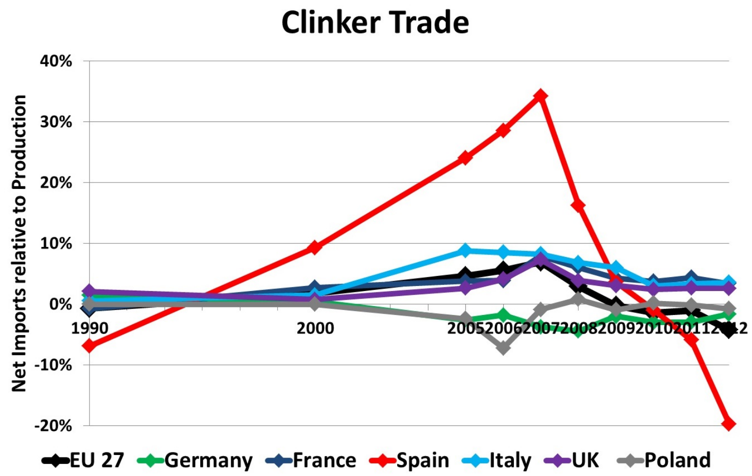

Figure 4.2: Net imports (imports minus exports) of clinker relative to local clinker production. Sources: Eurostat for net imports, EUTL and WBCSD GNR Database for production

The purpose of this chapter is to assess the actual emissions reductions in the cement industry, and not to provide a technology roadmap. Therefore just because clinker outsourcing is an undesirable option does not mean that it should not be considered in this context. Under the EU ETS, it can be profitable since, provided that a certain level of activity is maintained, the operator of an installation keeps receiving free allowances that can be sold on the market. However, logistical difficulties, high transportation costs and export barriers make clinker outsourcing less appealing than it appears. Clinker trading primarily occurs in the case of over- or under-capacity (Cook, 2011). Geography plays an important role: high road transport costs exclude inland producers from international trade (Ponssard and Walker, 2008).

Figure 4.2 shows clinker net imports (imports minus exports) divided by clinker production. The EU 27 switched from being a clinker importer to being a clinker exporter in 2009. We can see that clinker is a poorly traded commodity: since 1990 net extra-EU27 imports or exports have never been more than 7% of its production. Imports came from Asia (mostly China and Thailand) and the East Mediterranean region especially between 2001 and 2005 (mainly Turkey and Egypt), and since 2010 European clinker has mainly been exported to the Gulf of Guinea

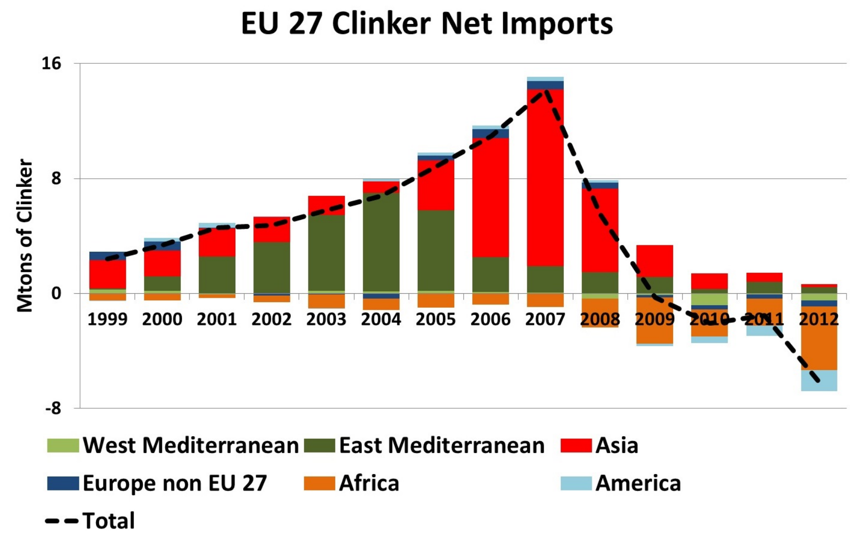

Figure 4.3: Origin of the EU 27 net imports. West Mediterranean comprises Morocco, Algeria, Tunisia and Libya. Source: Eurostat

and Brazil (see Figure 4.3). The European country with the most remarkable trajectory is Spain, which turned into a massive clinker exporter (20% of its production in 2012) after being a massive importer (up to 34% of its production in 2007). This swing can be explained by the boom and burst of the construction bubble. Further, most of the surge of clinker exports in 2012 compared to 2011 can be attributed to phase III allocation rules[13] (Chapter 5).

4.1.5 Alternative fuel use

The conventional fossil fuels used in clinker manufacture, coal and petcoke, have a high carbon intensity. Replacing these fuels by alternative, less carbon intensive fuels generates abatement. The proportion of alternative fuel used in thermal energy production has increased steadily in the European Union. Fossil and mixed wastes,[14] which are generally less carbon-intensive than coal or petcoke, repre-

[13] Allowances are cut by half if the plant produces less than half of its historical activity level. This encouraged plants to overproduce to reach the threshold. Excess clinker production has then been exported or blended in cement, increasing the clinker-to-cement ratio.

[14] Mostly plastics, mixed industrial waste, and tyres in 2012 (respectively 43%, 20% and 17% (source: GNR database, variable 3211detail).

sented 2% of thermal energy in 1990, 11% in 2005 and 25% in 2012.[15] Biomass represented[16] 0.2% of thermal energy in 1990, 4% in 2005 and 11% in 2012.[17] Most cement companies receive a fee for the burning of waste as part of a waste management strategy to reduce incineration and landfilling; so using alternative fuel may be financially advantageous regardless of the carbon price.

The carbon intensity of the fuel mix (shown in Figure 4.4) has decreased from 94 kgCO_2/GJ in 1990[18] to 80kgCO_2/GJ in 2012. In 2012, Germany had the lowest carbon intensity of the fuel mix by far (71 kgCO_2/GJ), while Italy had the highest (89 kgCO_2/GJ).

Much higher substitution rates are possible than the currently-used mixes but several factors limit the potential of alternative fuel use. First, the calorific value of most organic material is relatively low, and treatment of side products (such as chlorine) is sometimes needed (European Commission, 2010). Second, the availability of waste is dependent on the local waste legislation and collection network as well as nearby industrial activity (IEA, 2009). Third, a higher CO_2 price may increase the global demand for biomass, for which cement companies compete with heat and electricity producers. This would increase its price and make it less appealing as a fuel substitute for the cement industry. Finally, social acceptance is of huge relevance as incineration is often viewed with great suspicion by surrounding inhabitants.

4.1.6 Thermal and electrical energy efficiency

Cement manufacture requires both thermal energy for heating the clinker kiln and electrical energy (about 10% of total energy needed) mostly for kiln operation, grinding (preparing raw materials) and blending (mixing clinker with additives). The proportion of total electrical energy used for these steps is respectively 25%, 33% and 30% according to Schneider et al. (2011).

[15] GNR database, variable 3211a.

[16] Mostly animal meal and dried sewage sludge: respectively 49% and 20% (source: GNR database, variable 3211detail).

[17] GNR database, variable 3211a.

[18] For this value only, we took the average of European country values weighted by their cement production. Indeed, the original GNR value (91 kgCO_2/GJ) was lower than all values corresponding to individual European countries .

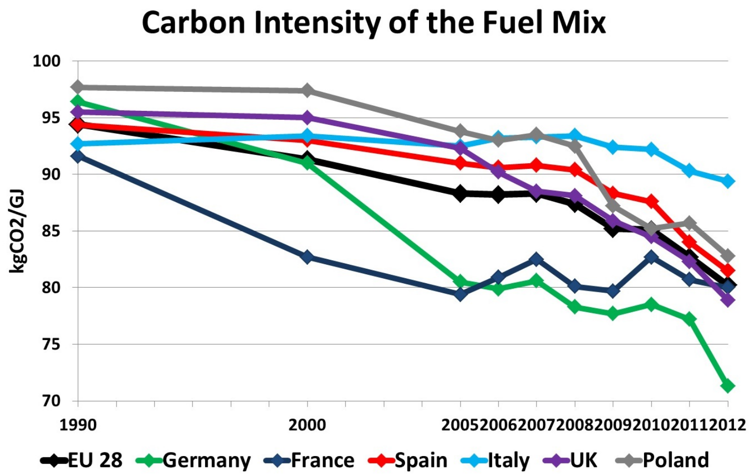

Figure 4.4: Carbon intensity of the fuel mix (in kgCO_2/GJ) for the EU 28 and main European countries. Source: WBCSD GNR Database, variable 3221

New kilns using raw material in powder form (dry production route) are much more energy efficient than old kilns using raw material in a slurry (wet production route) since less heat is needed to dry the raw material[19] (3-4 GJ per ton of clinker instead of 5-6 GJ per ton of clinker in European Commission (2010)). In modern kilns, part of the heat of the exhaust gases from the kiln is recovered to pre-heat the raw material (pre-heaters) (Pardo et al., 2011). The state-of-the art technology is the dry process kiln with pre-heating and pre-calcining, which requires approximately 3 GJ per ton of clinker and accounts for 46% of European clinker production in 2012 (compared to 23% in 1990[20]).

In addition to kiln technology, kiln capacity also influences energy efficiency. Bigger kilns have lower heat losses per unit of clinker produced and are therefore more energy-efficient. Finally, for a given installation, the way the machinery is operated (minimizing kiln shutdowns and operating near to nominal capacity) can make a significant difference (about 0.15-0.3 GJ per ton of clinker according to Hoenig and Twigg (2009)).

Cement producers benefit directly from energy efficiency through lower energy costs, which represent roughly a third of production costs (Bolscher et al.,

[19]It is common in the literature to distinguish four routes for cement manufacture: dry, semi-dry, semi-wet and wet (GNR).

[20]Source: GNR database, variable Percent315.

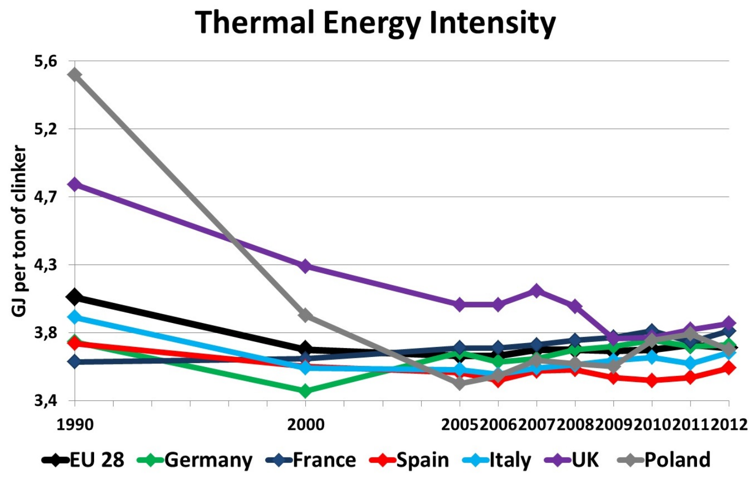

Figure 4.5: Thermal energy intensity in GJ per ton of clinker for the EU 28 and main European countries. Source: WBCSD GNR Database, variable 329.

2013; Pardo et al., 2011). Generally, new manufacturing plants are equipped with the best available technology, but the upgrading of old facilities is a slow process. Moya et al. (2011) find that the observed rate of retrofitting in the cement industry is much lower than the theoretical rate derived from the number of feasible improvements with low payback periods, revealing an "energy efficiency gap" (Jaffe and Stavins, 1994) or "energy efficiency paradox" (deCanio, 1998).

Figures 4.5 and 4.6 show the thermal energy intensity and the electrical energy intensity, respectively in GJ per ton of clinker and in kWh per ton of cement. The thermal energy intensity in the EU 28 decreased from 4.1 GJ per ton of clinker in 1990 to 3.7 GJ per ton of clinker in 2005 then stabilized. The electrical energy intensity in the EU 28, after decreasing from 114 kWh per ton of cement in 1990 to 108 kWh per ton of cement in 2006 increased to 116 kWh per ton of cement in 2012. The most noticeable change comes from Spain where average electricity intensity soared from 98 kWh/ton of cement in 2006 to 150kWh per ton of cement in 2012, probably due to the decrease in production which led to the use of machinery operating well below nominal capacity.[21]

[21] Spanish cement production was divided by three in the same period. To understand such a dramatic increase, whereas in the same time thermal energy intensity has not evolved, two explanations can be proposed. First, contrary to kiln fuel which is fully variable, there is a higher share of "non-productive" electric energy (such as lighting, heating, water pumping and compressed air).

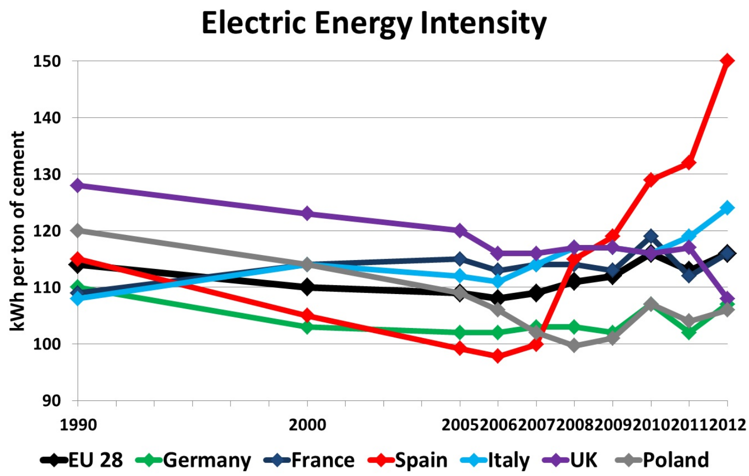

Figure 4.6: Electrical energy intensity in kWh per ton of cement for the EU 28 and main European countries. Source: WBCSD GNR Database, variable 3212.

There are no breakthrough technologies in sight that would allow a significant decrease in kiln energy consumption (European Commission, 2010), so the potential for abatement is small. In addition, the other abatement drivers can be negatively correlated to energy efficiency. Clinker substitutes (especially blast furnace slag) generally require more energy for grinding, and alternative fuels may provide less calorific power or may need more energy to treat by-products. Moreover, more stringent environmental requirements (dust and gas treatment), increased cement performance (necessitating finer grinding) and kiln improvements such as pre-heaters and pre-calciners have led to higher power consumption (Hoenig and Twigg, 2009). These reasons could explain why energy efficiency has stabilized or deteriorated in recent years.

4.1.7 Decarbonisation of electricity

For the sake of simplicity in this study we consider that all the electricity consumed comes from the grid.[22]. In this context, this mitigation option does not depend on

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Second, some plants have several kilns, so production can be redirected to the most efficient ones only. Conversely, most plants have only one grinder.

[22]The number of plants recovering heat for power generation is unknown (Matthes et al., 2008) Self-generation of power is more frequent in countries where electricity supply is not reliable (VDZ (German Cement Association), 2013).

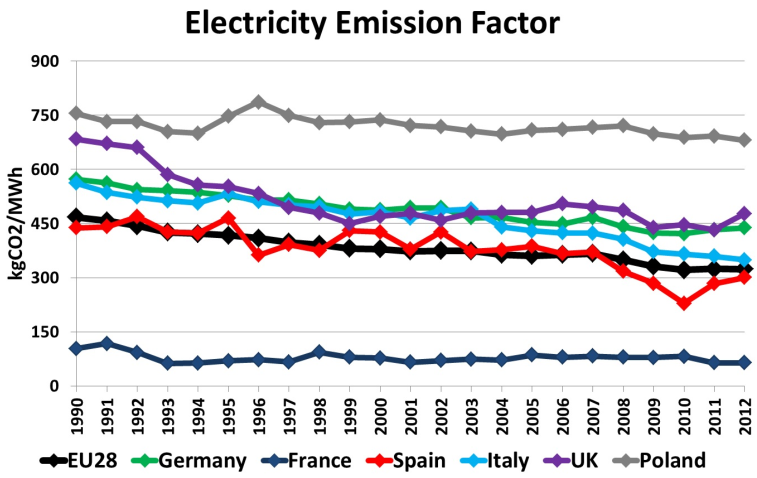


Figure 4.7: Electricity emission factor (in $kgCO_2/MWh$) for the EU 27 and main European countries. Source: Enerdata database

the cement industry but on electricity producers. Indirect electricity emissions represent around 6% of total emissions in the cement industry. Under the EU ETS framework, these emissions are attributed to electricity producers and not to cement manufacturers. Cement companies do not receive allowances for these emissions and neither do they have to surrender allowances for them. However, they may face indirect costs through the rise in electricity prices due to the passing-through of allowance prices. Though small, this abatement option still has the potential to decrease total emissions in the cement industry.

Figure 4.7 shows the changes in the electricity emissions factor (in $kgCO_2/MWh$). It has globally decreased in all European countries, and the EU 27 average dropped from 474 $kgCO_2/MWh$ in 1990 to 339 kg CO_2/MWh. In 2012, the country with the highest electricity emissions factor was Poland with 680 $kgCO_2/MWh$ (because of the predominance of coal power) and the country with the lowest was France with 69 $kgCO_2/MWh$ (because of the high proportion of nuclear and hydro-electric power).

4.1.8 Carbon capture and storage

Most carbon emissions from cement manufacturing are process emissions due to the chemical reaction during limestone calcination. The only way to avoid these emissions (apart from alternative cements based on different chemical processes) would be carbon capture and storage (CCS) using post-combustion technologies. Emissions due to burning of fossil fuels could also be managed with CCS technologies. A promising option in this direction is oxyfuel technology where air is replaced by oxygen in cement kilns to produce a pure CO_2 stream that is easier to handle (Barker et al., 2009; Li et al., 2013).

R&D in CCS is active but these potentially promising technologies are far from being operational at the industrial scale (Moya et al., 2010). A high carbon price (estimations vary but an order of magnitude is 50€/tonCO_2) would be necessary to trigger investments in this medium-term option. Furthermore, CCS technologies are energy-intensive and would increase power consumption significantly (by 50% to 120% at plant level according to Hoenig and Twigg (2009)). Finally, their large-scale development would necessitate a complete CCS system, including transport infrastructure, access to storage sites, a legal framework for CO_2 transportation, monitoring and verification, and therefore political and social acceptance (IEA, 2009).

4.1.9 Innovative cements

Several low-carbon or even carbon-negative cements are at the development stage, such as Novacem (based on magnesium silicates rather than limestone), Calera or Geopolymer (Schneider et al., 2011). Providing they prove their economic viability and gain customer acceptance (which is extremely challenging in itself), replacing existing facilities would require considerable time and investment.

4.1.10 Cement substitution in construction

This option, aimed at reducing the overall quantity of cement produced, depends on architects and construction companies. Like decarbonisation of electricity, it depends on other stakeholders. Whereas cement companies are indifferent to the carbon content of electricity (for a given electricity price), a reduction in quantities of cement used in construction is at first sight against the interests of the cement industry.

Reducing quantities of cement used in construction would be possible through alternative materials and/or leaner structures. Wood would be the most natural alternative construction material to cement, provided that its large-scale availability could be assured.

4.2 Methodology

Thus far, we have presented the emission abatement options qualitatively or on the basis of simple indicators. Quantifying their respective contribution in the evolution of cement CO_2 emissions requires a decomposition method, which we describe in the next section.

4.2.1 Decomposition of carbon emissions due to cement production

In the rest of this section, C stands for emissions, Q for quantities and E for energy consumption. The definition of all the variables used can be found in Table 4.2.

We distinguish $Q^{PROD}_{clinker,t}$ which is the quantity of clinker *produced* at year t and $Q^{NET}_{clinker,t}$ which is the quantity of clinker actually *used* for cement manufacture. The difference between the two comes from international trade (we neglect stock variations):

$$Q^{NET}_{clinker,t} = Q^{PROD}_{clinker,t} + NI_{clinker,t} \tag{4.1}$$

$NI_{clinker,t}$ being net imports of clinker. We split emissions into three categories: emissions due to fuel burning (subscript F), process emissions (subscript P) and indirect emissions due to electricity consumption (subscript E):

$$C_t = C_{F,t} + C_{P,t} + C_{E,t} \tag{4.2}$$

Only direct emissions are accounted for in the EU ETS.

$$C_{EUTL,t} = C_{F,t} + C_{P,t} \tag{4.3}$$

Table 4.2: Definition of variables

Variable	Definition	Unit
t	Year. All variables (except CEF_{pro}) are yearly	
C_t	Total carbon emissions in the cement manufacturing process	$MtonCO_2$
$C_{EUTL,t}$	Direct carbon emissions in the cement manufacturing process	$MtonCO_2$
$C_{F,t}$	Fuel-related emissions	$MtonCO_2$
$C_{P,t}$	Process emissions	$MtonCO_2$
$C_{E,t}$	Indirect carbon emissions due to electricity consumption	$MtonCO_2$
$Q^{NET}_{clinker,t}$	Quantity of cement manufactured	Mtons
$Q^{PROD}_{clinker,t}$	Quantity of clinker manufactured	Mtons
$NI_{clinker,t}$	Net imports (imports minus exports) of clinker	Mtons
$Q_{cement,t}$	Quantity of clinker used to manufacture cement	Mtons
H_t	Clinker home production ratio $(= \frac{Q^{NET}_{clinker,t}}{Q^{PROD}_{clinker,t}})$	None
R_t	Clinker-to-cement ratio	None
$I_{T,t}$	Thermal energy intensity	GJ per ton of clinker
$I_{El,t}$	Electrical energy intensity	MWh per ton of cement
$CEF_{F,t}$	Carbon intensity of the fuel mix	tCO_2/GJ
CEF_{pro}	Carbon emission factor of limestone calcination	tCO_2 per ton of clinker
$CEF_{elec,t}$	Electricity emission factor	tCO_2/MWh
A_t	Allocation cap	$MtCO_2$
C_eCI_t	Cement carbon intensity	tCO_2 per ton of cement
C_kCI_t	Clinker carbon intensity	tCO_2 per ton of clinker

Note: Some variable units for LMDI analysis differ from the units used in section 4.1.1

First, emissions due to fuel burning, $C_{F,t}$, can be decomposed as follows:

$$\begin{aligned} C_{F,t} =& Q_{cement,t} \frac{Q^{NET}_{clinker,t}}{Q_{cement,t}} \frac{Q^{PROD}_{clinker,t}}{Q^{NET}_{clinker,t}} \frac{E_{T,clinker,t}}{Q^{PROD}_{clinker,t}} \frac{C_{F,t}}{E_{T,clinker,t}} \\ &= Q_{cement,t} \times R_t \times H_t \times I_{T,t} \times CEF_{F,t} \end{aligned} \tag{4.4}$$

where $E_{T,clinker,t}$ is the thermal energy used, R_t the clinker-to-cement ratio, H_t is the clinker home production ratio ($H_t > 1$ if more clinker is produced than used, or, put another way, if net imports are negative), $I_{T,t}$ is the thermal energy intensity (in GJ per ton of clinker) and $CEF_{F,t}$ is the carbon intensity of the fuel mix (in tCO_2/GJ).

The formulation for process emissions $C_{P,t}$ is:

$$\begin{aligned} C_{P,t} =& Q_{cement_t} \frac{Q^{NET}_{clinker,t}}{Q_{cement,t}} \frac{Q^{PROD}_{clinker,t}}{Q^{NET}_{clinker,t}} \frac{C_{P,t}}{Q^{PROD}_{clinker,t}} \\ &= Q_{cement,t} \times R_t \times H_t \times CEF_{pro} \end{aligned} \tag{4.5}$$

where CEF_{pro} is the CO_2 emission factor for the calcination of limestone which is considered here time invariant, absent any information on its evolution.

The formulation for $C_{E,t}$ is:

$$\begin{aligned} C_{E,t} =& Q_{cement,t} \frac{E_{E,t}}{Q_{cement,t}} \frac{C_{E,t}}{E_{E,t}} \\ &= Q_{cement,t} \times I_{El,t} \times CEF_{elec,t} \end{aligned} \tag{4.6}$$

where $E_{T,clinker,t}$ is the electrical energy used, $I_{El,t}$ is the electrical energy intensity of production (in MWh per ton of cement) and $CEF_{elec,t}$ is the electricity emission factor (in tCO_2/MWh).

Total emissions of cement manufacturing are then

$$C_t = Q_{cement,t} \times (R_t \times H_t \times (CEF_{pro} + I_{T,t} \times CEF_{F,t}) + I_{El,t} \times CEF_{elec,t}) \tag{4.7}$$

Abatement levers are more visible in this formula that is composed only of positive terms: besides reducing activity (reducing $Q_{cement,t}$) or outsourcing clinker (reducing H_t), technological abatement options are reducing R_t (clinker substitution), $CEF_{F,t}$ (alternative fuel use), $I_{T,t}$ and $I_{El,t}$ (thermal and electrical energy efficiency), and reducing $CEF_{elec,t}$ (decarbonisation of electricity).

For the data, we have taken directly from GNR the intensity variables R_t (variable 3213), $CEF_{F,t}$ (variable 3221), $I_{T,t}$ (variable 329) and $I_{El,t}$ (variable 3212). These data are given at the EU 28 level (whereas we focus on the EU 27 level) but the error is low since they are intensity variables, and Croatia's cement production accounts for less than 2% of EU 28 cement production (Mikulčić et al., 2013). $CEF_{elec,t}$ comes from the Enerdata database and CEF_{pro} from Ecofys et al. (2009) (we take, unless explicitly mentioned otherwise, the measured value from the GNR database, 538 kgCO_2 per ton of clinker, rather than the default factor of 523 kgCO_2 per ton of clinker derived from the IPCC methodology used in the EU ETS).

H_t and $Q_{cement,t}$ are obtained indirectly by computation. The quantity of clinker produced is obtained by dividing EUTL emissions[23] by the clinker carbon intensity (using the EU ETS value of CEF_{pro}):

$$Q^{PROD}_{clinker,t} = \frac{C_{EUTL,t}}{CEF_{pro} + I_{T,t} \times CEF_{F,t}} = \frac{C_{EUTL,t}}{C_k CI_t} \tag{4.8}$$

where $C_k CI_t$ is the clinker carbon intensity. Then H_t is given by:

$$H_t = \frac{Q^{PROD}_{clinker,t}}{Q^{PROD}_{clinker,t} + NI_{clinker,t}} \tag{4.9}$$

where $NI_{clinker,t}$ comes from the Eurostat international trade database. $Q_{cement,t}$ is obtained by:

$$Q_{cement,t} = \frac{Q^{PROD}_{clinker,t} + NI_{clinker,t}}{R_t} \tag{4.10}$$

[23]Sometimes EUTL emissions do not exist (before 2005) or are not reliable: for the EU 27 in phase I, because some countries were not covered, and for the UK in phase I, because of the opt-out condition, some plants were not part of the scheme. In these cases we use GNR direct emissions, corrected by a factor to take into account the discrepancy between GNR and EUTL emissions. The factor is 2005-2010 EUTL emissions divided by 2005-2010 GNR emissions (we take the period 2008-2010 for the EU 27 and the UK).

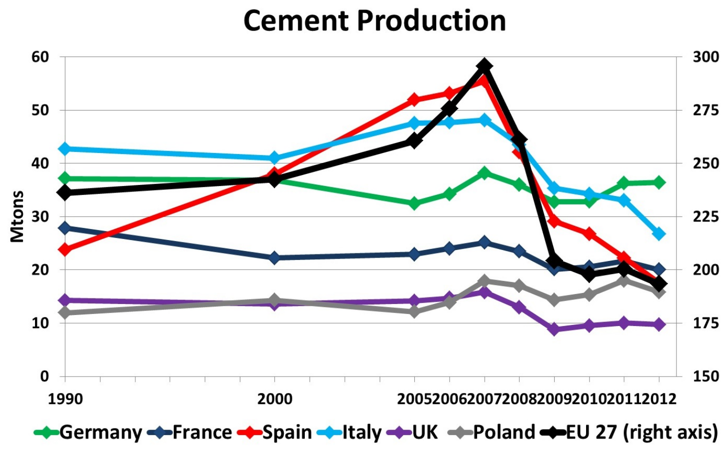


Figure 4.8: Cement production in million tons for the EU 27 (right vertical axis) and the main European countries (left vertical axis). Source: Computation from WBCSD GNR Database, EUTL database and Eurostat International Database

Computing indirectly clinker and cement production (whereas they are available in the GNR database) is a modelling choice. Indeed, equation (4.7) is a perfect accounting equality, but in practice there are always mismatches due to data inaccuracy, and so one variable has to be computed through the equation (instead of coming from data source). The choice of which variable to compute is determined by the quality of the data and the use of the decomposition. In our case we have a choice between using GNR data on clinker and cement production and compute emissions, or using EUTL emissions data for direct emissions and compute clinker and cement production. We chose the second option for two reasons. First, it allows finding EUTL emissions after recalculation for direct emissions, and EUTL emissions are extremely reliable: the coverage is 100%, and it comes from a compulsory policy rather than a voluntary program. GNR coverage is good but not perfect (some clinker plants are missing and grinding plants using imported clinker may not be covered). As an example, in Spain in 2007 (the country-year with the highest clinker importation), the GNR database gives a production of 46.8 Mt of cement, whereas our own computation (with 11.0 Mt of clinker net imports) gives 55.4 Mt, which are closer to the official figure of the Spanish cement association: 54.7 Mt (Oficemen, 2013). Second, it allows decomposing the *exact* allowances surplus and not an approximation of it (see section 4.4.3). Anyway, the difference between computed clinker production and reported GNR data at the EU level is small and stable over time.[24] So, given the order of magnitude of changes in the overall production, and because we are more interested in relative changes than absolute values, using one or the other would have hardly any impact on results of section 4.3.

4.2.2 LMDI method

Index decomposition analysis (IDA) has been widely used in studies dealing with energy consumption since the 1980s and carbon emissions since the 1990s. Ang (2004) compares different IDA methods and concludes that the Logarithm Mean Divisia Index (LMDI) is to be preferred. A comprehensive literature survey reviewing 80 IDA studies dealing with emissions decomposition is given in Xu and Ang (2013), and shows that the LMDI became the standard method after 2007.

The general formulation of LMDI (see Ang (2005)) is the following. When emissions can be decomposed as $C_t = X^1 \times X^2 \times \cdots \times X^n$, the variation of emissions $\Delta^{tot} = C_T - C_0$ can be decomposed as $\Delta^{tot} = \Delta^1 + \Delta^2 + \cdots + \Delta^n$, with

$$\Delta^k = \frac{C_T - C_0}{\ln(C_T) - \ln(C_0)} \times \ln(\frac{X_T^k}{X_0^k}) \tag{4.11}$$

LMDI decomposition is mostly used to study the difference in emissions between two dates for a given country, but the mathematical formulation also works for difference in emissions for two countries at a given date, or (as we will see later), for difference in emissions for a given country between a real and a counterfactual or reference scenario.

Among the 34 studies since 2002 using LMDI decomposition analysis in Xu and Ang (2013) literature review, the majority (14) are economy-wide and only seven focus on industry. But except for Sheinbaum et al. (2010) (iron and steel

[24] Computed production is higher than GNR data by about 3-4% for clinker and 6-7% for cement. Two reasons could explain the difference: the coverage (as coverage is around 95%, GNR underestimate cement production by about 5%) and white clinker. White clinker, which represents a tiny fraction of clinker production, is more carbon-intensive (by around 30%) than grey clinker. GNR intensity variables mostly concern grey clinker only, whereas EUTL emissions do not distinguish grey and white clinker. This introduces an upward bias in computed production. The higher the proportion of white clinker, the higher the bias; and the bias is stable in time if the proportion of white clinker production remains stable.

in Mexico), they are not sector-specific but deal with industry or the manufacturing sector as a whole; in China (Liu and Ang, 2007; Chen, 2011), Shanghai (Zhao et al., 2010), Chongqin (Yang and Chen, 2010), the UK (Hammond and Norman, 2012) or Thailand (Bhattacharyya and Ussanarassamee, 2004). For sector specific studies (not using the LMDI method), one can cite two international comparisons for cement (Kim and Worrell, 2002a) and steel (Kim and Worrell, 2002b) and a study of the iron and steel industry in Mexico (Ozawa, 2002).

The study closest to ours is Xu et al. (2012), which was not cited in Xu and Ang (2013), focusing on the cement industry in China. They give a decomposition per kiln type, allowing the energy efficiency effect to be separated into a structural effect (change of kiln type) and a kiln efficiency effect.[25] However they do not consider clinker trade in their decomposition, which is arguably of little importance for China, but matters for Europe.

Expanding equation (4.7) leads to the following decomposition:

$$\begin{aligned}
\Delta^{tot} &= C_T - C_0 \\
&= \triangle^{act-F} + \triangle^{sha-F} + \triangle^{tra-F} + \triangle^{fmix} + \triangle^{eff-F} \\
&\quad + \triangle^{act-P} + \triangle^{sha-P} + \triangle^{tra-P} \\
&\quad + \triangle^{act-E} + \triangle^{eff-E} + \triangle^{Celec} \\
&= \triangle^{act} + \triangle^{sha} + \triangle^{tra} + \triangle^{fmix} + \triangle^{eff-F} + \triangle^{eff-E} + \triangle^{Celec}
\end{aligned} \tag{4.12}$$

performing the appropriate groupings: $\triangle^{act} = \triangle^{act-F} + \triangle^{act-P} + \triangle^{act-E}$, $\triangle^{tra} = \triangle^{tra-F} + \triangle^{tra-P}$ and $\triangle^{sha} = \triangle^{sha-F} + \triangle^{sha-P}$.

The precise formulas are all given in the Appendix 4.6.1.

There are then seven factors in the decomposition:

- The *activity effect* ($\triangle^{act}$): impact of total cement production on emissions variations. It corresponds to lever (i) in part 4.1.1

- The *clinker trade effect* ($\triangle^{tra}$): impact of the clinker trade on emissions variations. It corresponds to lever (iii) in part 4.1.1.

[25] Kiln energy intensity over time per kiln type was not available in the GNR database, so we opted for a simpler decomposition.

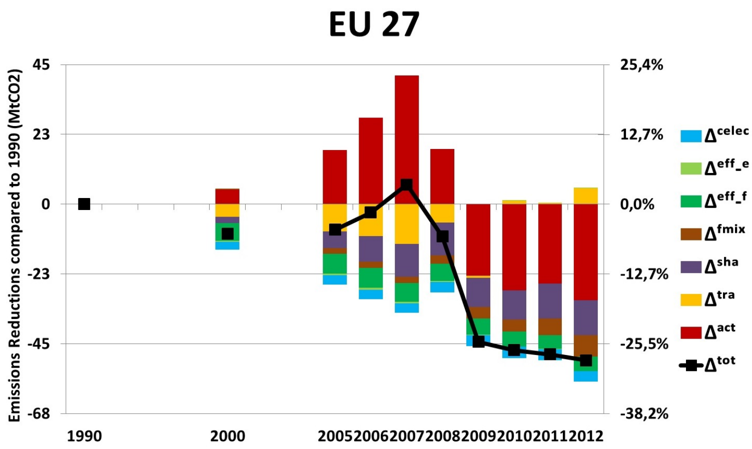


Figure 4.9: LMDI decomposition analysis of cement emissions compared to 1990. EU 27

- The *clinker share effect* ($\triangle^{sha}$): impact of clinker substitution on emissions variations. It corresponds to lever (ii) in part 4.1.1.

- The *fuel mix effect* ($\triangle^{fmix}$): impact of the use of alternative fuel on emissions variations. It corresponds to lever (iv) in part 4.1.1.

- The *thermal* and *electrical energy efficiency effect* ($\triangle^{eff-F}$ and $\triangle^{eff-E}$): impact of thermal and electrical energy efficiency. They correspond to lever (v) in part 4.1.1.

- The *electricity carbon emissions factor effect* ($\triangle^{Celec}$): impact of the carbon emissions factor on emissions variations. It corresponds to lever (vi) in part 4.1.1.

One can distinguish the first two effects (activity and clinker trade) which are "non-technological" abatement options from the others that are technological abatement options.

4.3 Changes in carbon emissions in the European cement industry

4.3.1 EU 27

Figure 4.9 shows changes in carbon emissions over time compared to their 1990 level alongside the LMDI decomposition analysis explained above.[26]

Emissions in the cement industry first decreased in the 1990s and the beginning of the 2000s (-4.7% from 1990 to 2005) then increased sharply to exceed the 1990 level (+3.6% in 2007 compared to 1990). The economic recession led to a sharp decrease in emissions: in 2009 they were 25.1% lower than in 1990 (which corresponds to a 29.1% reduction in emissions in two years) and kept decreasing slowly afterwards.

The LMDI analysis allows us to highlight the fact that most of the emissions variations in the EU 27 are attributable to the activity effect: cement emissions have increased or decreased mostly because more or less cement has been produced. The activity effect was responsible for an increase of 41.5 Mtons of CO_2 in 2007 compared to 1990 (+22.7%) and for a decrease of 64.5 Mtons of CO_2 two years later (corresponding to a 34.2% decrease).

At the European level, the clinker trade effect partially compensates the activity effect most of the time: it is negative when the activity effect is positive and vice-versa. Put differently, a production increase is accompanied by an increase in clinker net imports and a production decrease by a decrease in clinker net imports, which can be explained by production capacity constraints (Cook, 2011). Keeping 1990 as the reference level, the clinker trade effect was at its highest in 2007 when clinker net imports reached 14.1 Mtons. At this time, 12.8 Mtons of CO_2 (7% of 1990 emissions) were avoided in Europe because of clinker outsourcing. With the economic downturn and the decrease in overall production, clinker net imports dropped and Europe became a clinker exporter in 2009. Between 2007 and 2010, while the activity effect led to a decrease of 69.2 Mtons of CO_2, the change in the balance of the clinker trade was responsible for an increase of 13.9 Mtons of CO_2 in Europe.

[26] In the graphic we display variations from 1990 (fixed date) to year i. To compute variations between years i and j, we only have to take the differences, as the decomposition is linear and $\triangle_{i,j} = C_i - C_j = C_i - C_{1990} - (C_j - C_{1990}) = \triangle_{i,1990} - \triangle_{j,1990}$.

The two most important levers of technological emissions reduction are clinker substitution and alternative fuel use. The clinker share effect led to a reduction of 5.4 Mtons of CO_2 in 2005 compared to 1990 (-3.0%) and an extra 5.9 Mtons in 2012 compared to 2005 (-3.4%). Alternative fuel use led to a reduction of 1.9 Mtons of CO_2 in 2005 compared to 1990 (-1.0%) and an extra reduction of 5.1 Mtons between 2005 and 2012 (-2.9%).

Thermal energy efficiency was the most important driver of emissions reduction in the 1990s: between 1990 and 2000, it induced a decrease of 5.7 Mtons of CO_2 (-3.2%). Since then, thermal energy efficiency in Europe has stagnated, generating no extra emissions reduction. The electrical energy efficiency effect has by far the least influence. It led to 0.5 Mtons of CO_2 of emissions reduction between 1990 and 2005. Then a deterioration in electrical energy efficiency led to an increase of 0.7 Mtons of CO_2 between 2005 and 2012. There are two possible explanations for the stagnation of thermal energy efficiency and the deterioration of electrical energy efficiency in the 2000s. First, kilns were operating below capacity and thus below their optimal efficiency level. Second, the two other main abatement options (clinker reduction and alternative fuel use) may reduce energy efficiency (see part 4.1.6).

Finally, the electricity carbon emissions factor effect has had a progressive impact in reducing cement emissions, globally small but not negligible. This channel of emissions reduction, which has the particular characteristic of depending on other stakeholders than the cement industry itself, was responsible for a decrease of 2.5 Mtons of CO_2 between 1990 and 2000 and 0.9 Mtons of CO_2 between 2000 and 2012 (-1.4% then -0.5%).

These observations can be summarised as follows. Clinker substitution, alternative fuel use, and to a lesser extent decarbonisation of electricity, have brought a continuous decrease in carbon emissions over the past twenty years (respectively 11.3, 7.0 and 3.3 Mtons of CO_2 between 1990 and 2012, i.e. 6.2%, 3.8% and 1.8% reduction). Together they are responsible for a 11.9% decrease in carbon emissions. Energy efficiency induced a decrease in emissions in the 1990s (5.7 Mtons of CO_2 or -3.2% between 1990 and 2000) then a small increase, probably because of clinker share reduction and alternative fuel use. Overall it was responsible for 4.7 Mtons of emissions reductions between 1990 and 2012 (-2.6%). Apart from

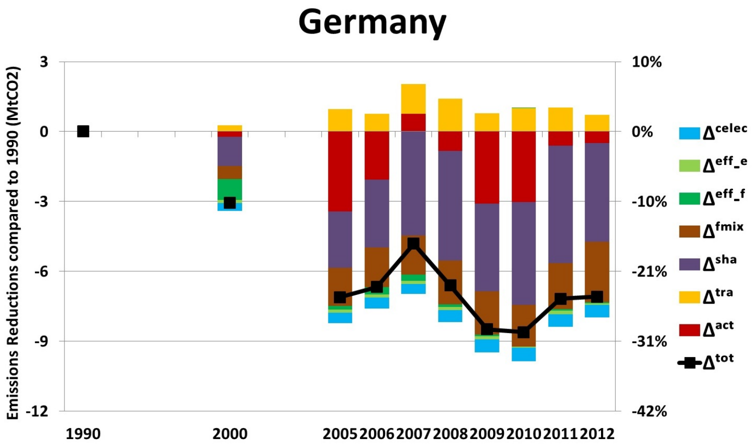


Figure 4.10: LMDI decomposition analysis of cement emissions compared to 1990. Germany

this long-time slow trend of emissions reduction, most of the emissions fluctuations are explained by the activity effect, which is partially compensated for by the clinker trade effect.

4.3.2 Main European producers

Figures 4.10 to 4.15 show information, using the same graphical format, for the biggest European cement producers: Germany, France, Spain, the UK, Italy and Poland. We do not give such a detailed analysis for each country as for the EU 27 but only highlight the most salient facts.

- *Germany*. Germany shows that it is feasible to decrease significantly emissions intensity. Clinker substitution and alternative fuel use have allowed significant emissions reductions (-23% between 1990 and 2012). Moreover, Germany was exporting clinker at the peak of economic activity in 2007 while EU 27 as a whole was importing it. It is the only big Western European country which did not have a sharp decrease in cement production. Cement production was only 2% lower in 2012 than in 1990, while carbon emissions were 27% lower.

- *France*. France reduced emissions while making virtually no technological improvement between 2000 and 2012. In the 1990s the cement French

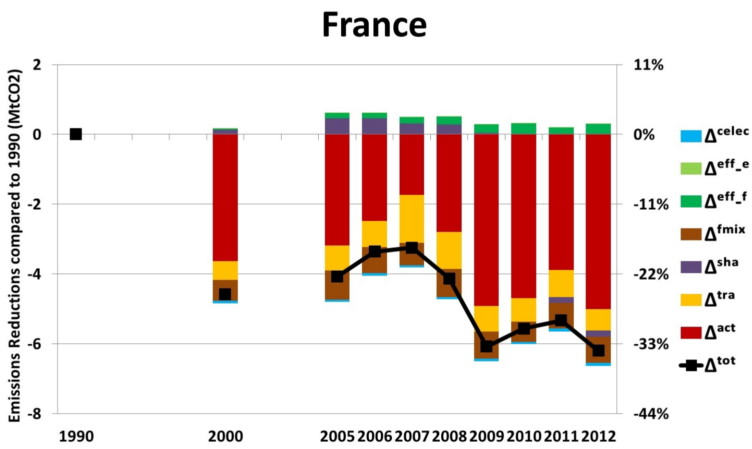


Figure 4.11: LMDI decomposition analysis of cement emissions compared to 1990. France

market got consolidated (only four companies were in activity in 2005, contrary to Germany, Italy and Spain where the market is more fragmented), which involved many plant closures. This could explain why the activity effect and the clinker trade effect did not move in opposite directions between 1990 and 2000 (the decrease in cement production, around 20%, by far more important than its European counterparts, was accompanied by a rise in clinker net imports). The clinker share effect, after being responsible for an increase in emissions until 2006, brought emissions reductions afterwards, returning approximately to its 1990 level, whereas in most European countries (except Italy) it has been a continuous source of significant emissions reductions. Energy efficiency, which was the best among the big Western European countries in 1990, has deteriorated continuously and led to an increase in emissions. The biggest source of emissions reduction, alternative fuel use, was only applied in the 1990s: hardly any improvement was achieved afterwards.

- *Spain*. Spanish cement emissions are overwhelmingly affected by the activity effect and the clinker trade effect. At the highest point of the housing bubble in 2007, the activity effect would have doubled emissions (+105%) compared to 1990, but was partially compensated for by the clinker trade

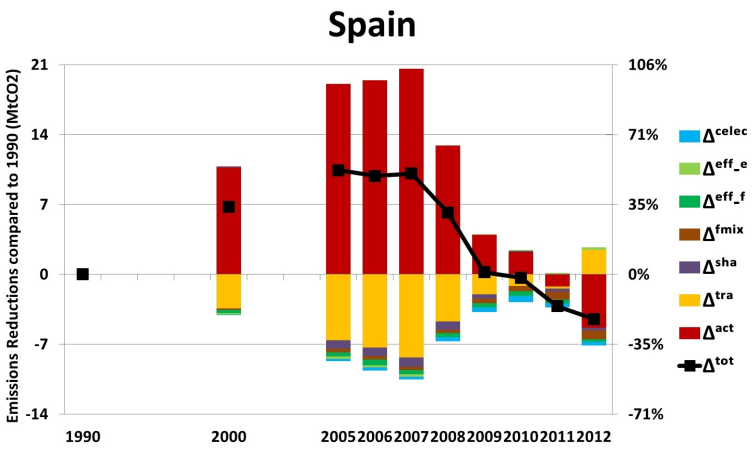


Figure 4.12: LMDI decomposition analysis of cement emissions compared to 1990. Spain

effect (-42%). The bursting of the housing bubble led to a massive reduction in cement production and therefore of emissions through the activity effect, which was partially offset by a massive reduction in clinker net imports, and an increase in the clinker-to-cement ratio. Still, some emissions reduction was achieved by alternative fuel use (especially since 2010), thermal energy efficiency, and electricity decarbonisation, bringing altogether 7.8% of emissions reductions in 2012 compared to 1990.

- *UK.* In 1990 the UK cement industry was the most CO_2-intensive in Western Europe. However, twenty years later it was one of the best performers. The reduction of the exceptionally high clinker-to-cement ratio (94% in 1990) down to 70% in 2012 led to massive emissions reductions (a 18% decrease compared to 1990). Other levers of emissions reduction such as energy efficiency and alternative fuel use were applied to a significant extent. On top of all these factors, the economic downturn led to a considerable decrease in emissions in 2008 and 2009 with a small rebound afterwards (whereas the activity effect was responsible for a small increase in emissions in 2005-2007). Overall, the UK is the major European country with the biggest fall in emissions in 2012 compared to 1990 (-58%, compared to -25% in Germany, -34% in France, -40% in Italy, -23% in Spain

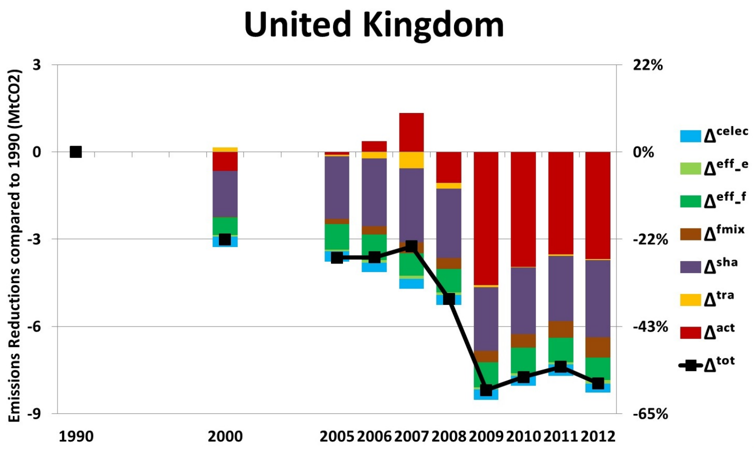


Figure 4.13: LMDI decomposition analysis of cement emissions compared to 1990. the UK

and -6% in Poland).

- *Italy*. Like France, Italy had good environmental indicators in 1990 such as the lowest clinker-to-cement ratio and a relatively low carbon intensity of the fuel mix. While being a major source of emissions reductions in other countries, the clinker share effect led to an increase in emissions in Italy, because of the increase in the clinker-to-cement ratio in the 1990s and its stabilization in the 2000s. Moreover, since 2000, barely any progress has been made in energy efficiency and alternative fuel use. The activity effect has had a qualitatively similar impact as in the UK (as Italy produces approximately twice as much cement, the effect is twice as small in percentage terms). Overall, the 40% emissions reduction compared since 1990 is almost entirely explained by the activity effect.

- *Poland*. Unlike the other European countries, Poland has had a sustained increase in production (only slightly hit by the recession). In 2012 the activity effect was responsible for a 27% increase in emissions compared to 1990. Most of this increase was compensated for by other sources of emissions reductions, explaining why emissions decreased by 6% in 2012 compared to 1990. The biggest contribution to emissions reduction was from energy efficiency, mostly in the 1990s, but clinker substitution and alternative fuel

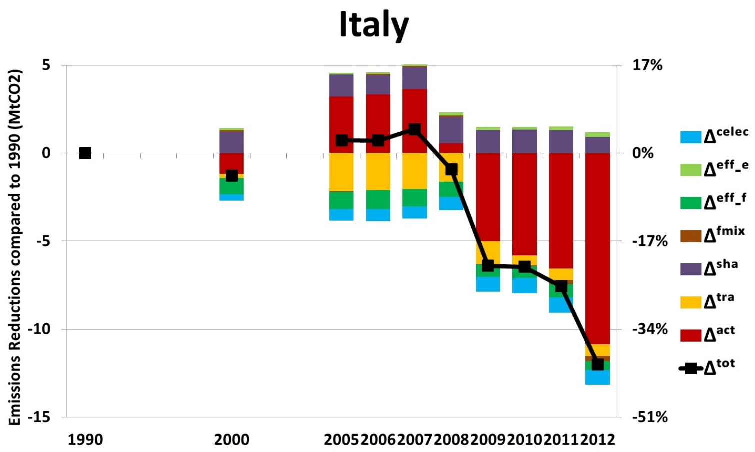


Figure 4.14: LMDI decomposition analysis of cement emissions compared to 1990. Italy

have had a significant impact.

4.4 Impact of the EU ETS on the cement industry

4.4.1 Overview

Figure 4.16 shows the results of the LMDI decomposition before (2000 and 2005) and after (2005-2012) the launch of the EU ETS.

Between 2000 and 2005, cement industry emissions increased by 0.7%, whereas between 2005 and 2012 they dropped by 24.9%. This gives the impression that the EU ETS was extremely efficient at reducing emissions. However, the LMDI analysis shows that the activity effect itself accounts for 25.9% of emissions reduction between 2005 and 2012, compensated for by a 7.2% increase in the clinker trade effect. This decrease in clinker net imports is essentially due to weak domestic demand leading to production overcapacity.

Among the technological abatement options, between 2005 and 2012, the clinker share effect, the fuel mix effect and the decarbonisation of electricity led to emission reductions of 3.8%, 2.9%, and 0.4% respectively, compensated for by a 0.4% increase due to the energy efficiency effect. Before the beginning of the EU ETS, between 2000 and 2005, the clinker share effect, the fuel mix effect, the carbon

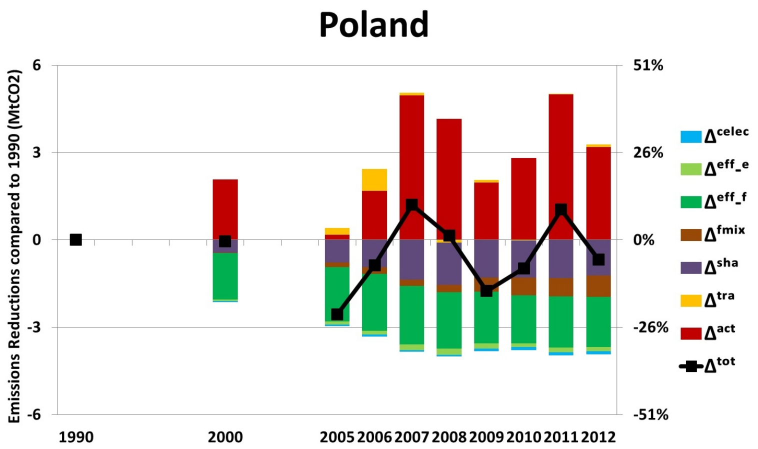


Figure 4.15: LMDI decomposition analysis of cement emissions compared to 1990. Poland

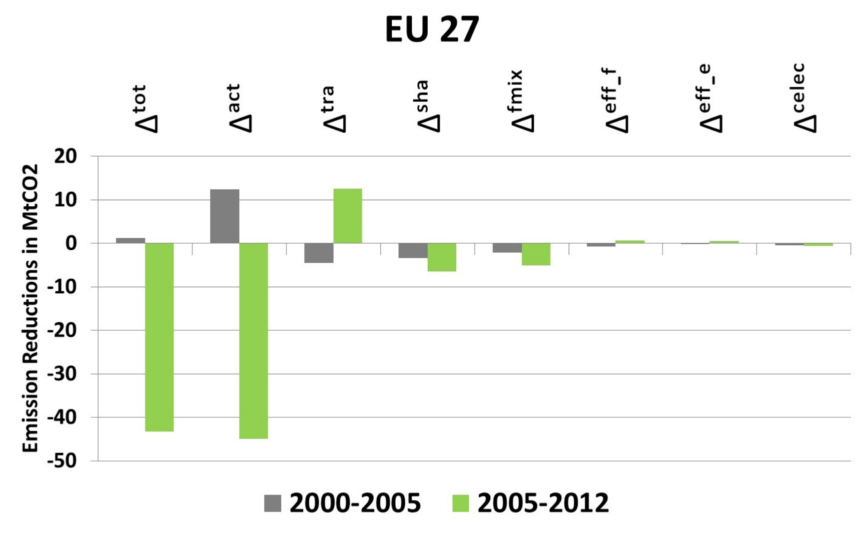


Figure 4.16: LMDI decomposition analysis of cement emission variations in the EU 27, before and after the beginning of the EU ETS (2000-2005 and 2005-2012)

emissions factor effect and the energy efficiency effect led to emissions reductions of 2.0%, 1.2%, 0.3%, and 0.4% respectively.

It would thus seem that the introduction of the EU ETS may have, to a small extent, accelerated the use of clinker substitution, alternative fuel use and decarbonisation of electricity,[27] while these mitigation options may have led to a decrease in energy efficiency.

Figure 4.16 does not show *abatement* but simply *changes* in emissions over time. Abatement is the difference between *actual* emissions and *counterfactual* emissions, which would have occurred if the EU ETS had not existed. Quantitative estimation of the abatement due to the EU ETS therefore necessitates the construction of a counterfactual scenario. The methodology and results are given in the next section.

4.4.2 Abatement

The method has three stages. First, we produce two counterfactual scenarios making assumptions about the different parameters of the emissions decomposition detailed in section 4.2.1. Second, we compute the difference $C_t^{real} - C_t^{counterfact}$ for each year, then decompose it through an LMDI decomposition analysis. Third, we add the different yearly effects and analyse the different levers of abatement. In this section and the next one, we consider the geographically changing EU ETS perimeter[28] instead of the EU 27, as we study the impact of the EU ETS on the cement industry.

For the counterfactual scenario, we assume that both the quantity of cement produced ($Q_{cement,t}^{counterfact} = Q_{cement,t}$), and the home production ratio ($H_t^{counterfact} = H_t$) remain unchanged. The EU ETS may have led to greater levels of production after the economic recession because of the allowance allocation method (which discourages plant closure); or conversely to lower levels of production because

[27] Though most of the decarbonisation of electricity may be due to renewable subsidies rather than the EU ETS itself (Weigt et al., 2013).

[28] The EU 27 minus Romania and Bulgaria until 2007, plus Norway, Lichtenstein and Iceland after 2008. However, data for Cyprus and Bulgaria are not available until 2008, and UK data are inaccurate for phase I because of the opt-out condition. The geographical perimeter considered at the European level is guided by the available EUTL data coverage (so only a part of UK production is considered in phase I). The production of clinker and cement as well as net imports have been modified to take into account the changing geographical perimeter.

of cement substitution (lever (i)), loss of competitiveness and leakage incentives (which have not been empirically proven so far, see Chapter 1); but these effects are likely to be small.

For the other variables, R_t, the clinker-to-cement ratio, $CEF_{F,t}$ the carbon intensity of the fuel mix, $I_{En,t}$, the thermal energy intensity of production, $I_{El,t}$, the electrical energy intensity of production and $CEF_{elec,t}$, the electricity emission factor; we consider two counterfactual scenarios. In the "Freeze" scenario, the variables keep their 2005 values from 2005 to 2012. In the "Trend" scenario, the variables decrease (or increase) at the same rate as the average yearly variation between 2000 and 2005.[29]

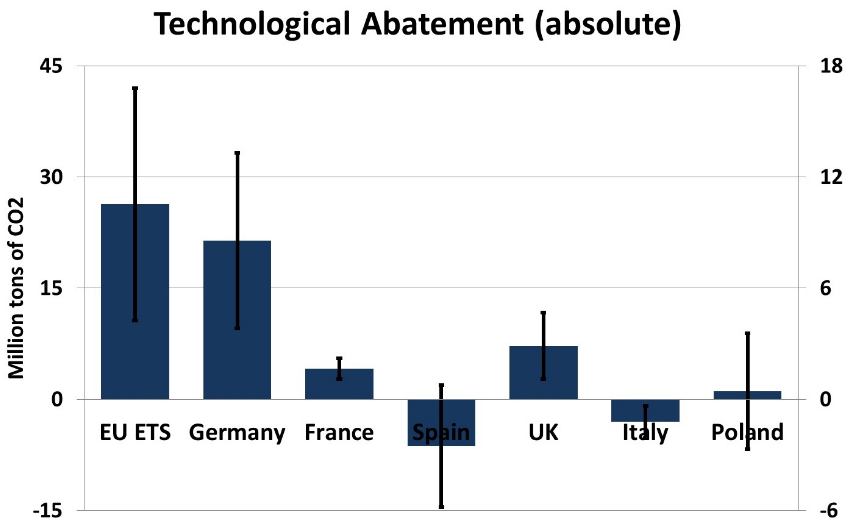
As an example, let us consider a given country for which the clinker-to-cement ratio is 80% in 2000 and 77% in 2005 (which corresponds to an average decrease of 0.8% per year). In the "Freeze" scenario, the clinker-to-cement ratio will stay at 77% from 2005 to 2012. In the "Trend" scenario, the clinker-to-cement ratio will start at 77% in 2005 and decrease by 0.8% per year, to finish at 73.5 % in 2012. In this case, the estimated abatement will be higher in the "Freeze" scenario, since the counterfactual scenario is more pessimistic (higher emissions).

Estimating what would have happened in the absence of an event (here the introduction of the EU ETS) is in itself very challenging. Suggesting that parameter values would have ranged between the "Freeze" and "Trend" scenarios is a rule of thumb that is admittedly simplistic, but has the virtue of avoiding the setting of arbitrary values for the parameters. Table 4.3 displays results when this method is applied for predicting 2005 values in the EU 28. Except for the clinker-to-cement ratio and the carbon intensity of the fuel mix, slightly out of the interval, the order of magnitudes are fairly correct.[30]

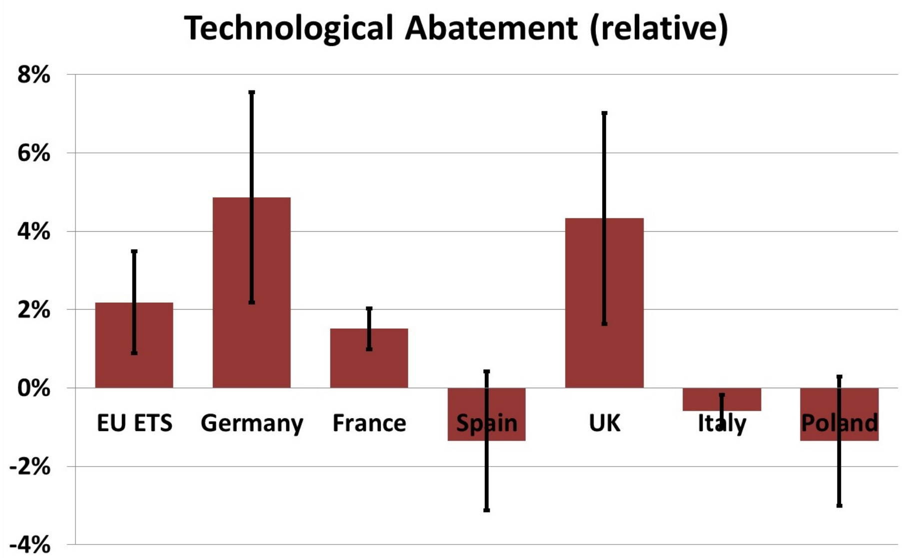
Figure 4.17 shows the results of the abatement estimates. Values shown correspond to the average of the two scenarios, and with the original values of scenarios

[29]Ideally we would have used the 2004 values if they had been available in the GNR database as in this method technological abatement is necessarily zero in 2005. However some time was probably needed for cement companies to adapt and take the EU ETS into account in their operational decisions.

[30]Our counterfactual is likely to be more precise for two reasons. First, the trend is based on a shorter term (5 years instead of 10). Second, 1990 was the first year for which data is collected, which was done in 2005. Then the level of assurance of 1990 details is not to the standard of later years.



(a) Absolute abatement (left axis for the EU ETS perimeter, right axis for the others) in millions of tons.



(b) Relative abatement as a percentage of total emissions

Figure 4.17: "Technological" abatement between 2005 and 2012. The bars correspond to the "Freeze" scenario estimates (the top bar except for France) and the "Trend" scenario estimates (the bottom bar except for France).

Table 4.3: Verification. Do the "Freeze" and "Trend" scenarios provide a good interval for changes in variables over time? Application on year 2005 for the EU 28 using 1990 and 2000 values. In this case, 2005 "Freeze" values are equal to 2000 values, and the trend rate is the one between 1990 and 2000.

Variable	Unit	1990	2005 "Freeze"	2005 Real	2005 "Trend"	Observation
R_t	%	78.4%	77.5%	75.9%	77.1%	Overestimation
$I_{En,t}$	GJ per ton of clinker	4,07	3,73	3,69	3,57	Interval OK
$I_{El,t}$	kWh per ton of cement	114	110	109	108	Interval OK.
$CEF_{F,t}$	$kgCO_2/GJ$	90.9	91.3	88.3	91.5	Overestimation
$CEF_{elec,t}$	$kgCO_2/MWh$	474	381	363	342	Interval OK

as the error interval. We find that between 2005 and 2012, the European cement industry abated 26 Mtons ($\pm$ 16 Mtons) of CO_2 emissions, which corresponds to a decrease of 2.2% ($\pm$ 1.3%) in emissions. However, this abatement could be due to an external cause - energy prices - rather than to the EU ETS. Indeed the prices of steam coal and petcoke (the two main energy sources used to produce clinker) roughly doubled from 2003-4 to 2010-11 as the graphs on p.31 of **?** show.[31] Increasing "conventional energy" prices reinforce the profitability of using substitutes rather than clinker, alternative fuels, and increasing energy efficiency.

Germany is the European country that has abated the most in absolute terms (9 Mtons $\pm$ 5 Mtons) and in percentage terms with the UK[32] (-4.9% $\pm$ 2.7% and -4.3% $\pm$ 2.7%). The abatement in France is small but positive (-1.5% $\pm$ 0.5%) while the abatement in Italy is small but negative (+0.6% $\pm$ 0.4%). The uncertainty in the evaluation of abatement in Spain and Poland is high (but both average values are negative).

The results described above come from a simple difference between actual and counterfactual emissions. An LMDI decomposition analysis allows us to investigate what levers have been used to provide actual abatement. The results are shown in Figure 4.18. Almost all of the technological abatement in the EU ETS perimeter comes from clinker share reduction (between 16 and 32 Mtons of CO_2)

[31] Steam coal and petcoke prices have fallen since 2011, due, among other reasons, to the shale gas boom in the US. If the downward trend persists, a degradation of the cement performance indicators would support this explanation.

[32] For the UK at the national level, we use corrected GNR data for emissions for phase I as in the previous section because of the inaccuracy of EUTL data.

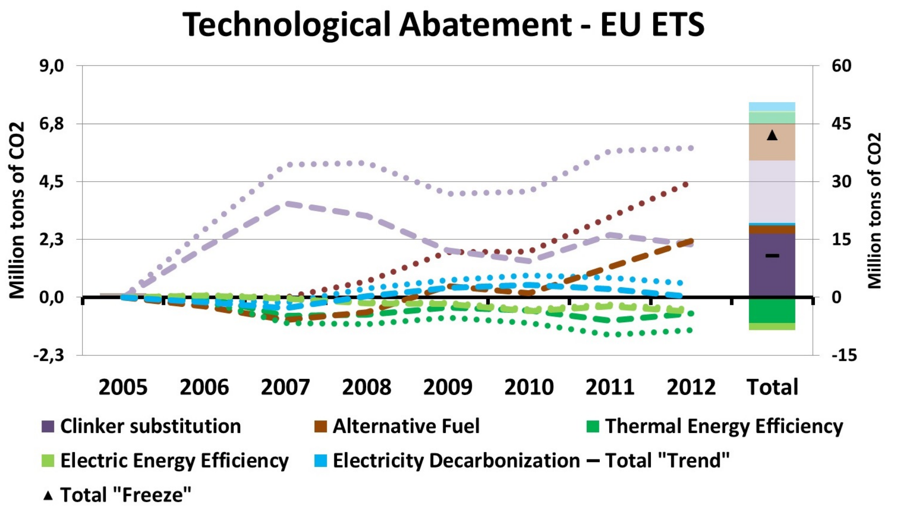


Figure 4.18: Technological abatement in the EU ETS perimeter. The curves on the left side show the abatement due to the different effects under the "Freeze" scenario (dotted line) and the "Trend" scenario (dashed line). The histogram on the right gives the sum of abatements over the years, in full color for the "Trend" scenario, and in full color *plus* faded color for the "Freeze" scenario.

followed by alternative fuels (between 2 and 12 Mtons of CO_2), while the decrease in energy efficiency led to negative abatement (between 4 and 7 Mtons of CO_2).

The detailed results, country by country, are given in the Appendix 4.6.2 and a summary of the results is given in Table 4.4. Clinker reduction is the main lever of technological abatement and led to actual abatement in Germany, France, the UK, and Poland but negative abatement in Spain and Italy. In all countries except France, abatement due to clinker substitution decreased (being negative in some countries) after the economic downturn. This could be explained by overcapacity and excess clinker production. Alternative fuel led to positive abatement in Spain, the UK and Poland and negative abatement in other countries (in France and Germany it could be because decarbonisation of the fuel mix had already started before the beginning of the EU ETS, so the "Trend" scenario gives lower emissions and actual abatement is harder to achieve). The thermal energy efficiency effect brought positive abatement in Germany, was neutral in the UK and brought negative abatement in France, Italy, and Poland. The electric energy efficiency effect brought positive abatement in the UK and Poland, was neutral in France and brought negative abatement in Germany, Italy and Spain. Electricity decarbonisation led to positive abatement in France, Spain, the UK and Italy and was insignificant in Germany and Poland.

Table 4.4: Impact of different technological options on technological abatement

	EU ETS	Germany	France	Spain	UK	Italy	Poland
Clinker reduction	+	+	+	−	+	−	+
Alternative fuel	+	−	−	+	+	=	+
Thermal Energy Efficiency	−	+	−	=	=	−	−
Electric Energy Efficiency	−	−	=	−	+	−	+
Elec Decarbonisation	+	=	+	+	+	+	=

Note: a + in clinker reduction means that clinker reduction indeed provided positive technological abatement. = stands for indeterminate (when the error interval overlaps zero in the decomposition)

4.4.3 Overallocation profits

Numerous studies have demonstrated that electricity companies have reaped windfall profits by passing through the allowance price to their consumers while they had received the allowances for free (Sijm et al., 2006). Indeed, even allocated free of charge, allowances can be sold and therefore have an opportunity cost.

The ability to pass through the allowance price to consumers has not been well-established for cement companies. Economic theory suggests that for linear demand curves, pass-through rates are higher in competitive markets than in monopolies (because prices are more directly linked to marginal production costs), and for markets with elastic supplies and inelastic demands (Sijm et al., 2008; Wooders et al., 2009). The cement industry is an oligopoly with moderately elastic supply and inelastic demand (Selim and Salem, 2010), which would suggest moderately-high pass-through rates (75-80%) (Oxera Consulting, 2004). To our knowledge, the only two empirical studies of pass-through rates in the European cement sector are to our knowledge an old study from Walker (2006), which unveils positive but moderate pass-through rates for 2005 (25-35%, depending on the country), and Alexeeva-Talebi (2010) for the German cement, lime and plaster sector finding a higher pass-through (73%).

In this chapter, we focus on another source of "windfall" profits obtained from the EU ETS: overallocation profits. The principle of overallocation profits is straightforward. When the number of EUAs,[33] given free of charge to cement companies, is higher than emissions necessary to manufacture the amount of cement really produced; a surplus of EUAs is automatically generated. These allowances can then be sold and will generate profits.

If we eliminate emissions due to electricity consumption from equation (4.7), which are not accounted for by cement companies in the EU ETS, we have the equation:

$$C_{EUTL,t} = Q_{cement,t} \times R_t \times H_t \times (CEF_{pro} + I_{En,t} \times CEF_{F,t}) \tag{4.13}$$

where $C_{EUTL,t}$ are direct emissions, $Q_{cement,t}$ is cement production, H_t the clinker home production ratio, CEF_{pro} the process emissions, $I_{En,t}$ the energy intensity and $CEF_{F,t}$ the carbon intensity of the fuel mix.

With the given state of technology in the EU 28 in 2005, and no clinker trade $(H = 1)$, an allocation cap A_t allows the production of a certain quantity of cement $Q^{A_t}_{cement}$ without buying or selling allowances:

$$\begin{aligned} Q^{A_t}_{cement} &= \frac{A_t}{R_{EU28_{2005}}(CEF_{pro} + I_{En,EU28_{2005}} CEF_{F,EU28_{2005}})} \\ &= \frac{A_t}{C_e CI_{EU28_{2005}}} \end{aligned} \tag{4.14}$$

with the cement carbon intensity of the EU 28 in 2005 $(C_e CI_{EU28_{2005}})$ being 656 kg of CO_2 per ton of cement.[34] In the rest of the chapter we will call $Q^{A_t}_{cement}$ the "production equivalent associated with the cap A_t".

We compute the difference between *actual* emissions C_t (associated with values $Q_{cement,t}$, R_t, H_t, $I_{En,t}$ and $CEF_{F,t}$) and the *reference* situation corresponding to the

[33] for the European Union Allowance, the "standard" allowance. Allowances from offset credits are CER (Certified Emission Reductions) for Clean Development Mechanisms and ERU (Emissions Reduction Units) from Joint Implementation.

[34] Calculated with the EU 28 values in 2005 of R, CEF_{pro}, I_{En} and CEF_F which are respectively 75.9%, 0.538 tCO_2/ton of clinker, 3.69 GJ/ton of clinker and 0.0883tCO_2/MJ.

cap A_t (associated with values $Q^{A_t}_{cement}$, $R_{EU28_{2005}}$, $H = 1$, $I_{En,EU28_{2005}}$ and $CEF_{F,EU28_{2005}}$); and decompose it[35] using the same LMDI decomposition method as in section 4.3. We then keep the activity effect, the clinker trade effect, and group the other effects under the name "technology" effect.

The technology effect gives the proportion of the EUAs surplus due to technological performance, while the activity and clinker trade effects give the proportions of the EUAs surplus due to underactivity and clinker outsourcing. Overallocation is then defined as the sum of the activity and clinker trade effects. The computed overallocation can be seen as the difference between actual allocation and output-based allocation, based on current clinker production with a certain level of technology (European average in 2005).

We choose to base the reference situation for technological performance on the 2005 European average values so that the reference situation brings *zero extra costs on average at the European level*. The estimation of overallocation is then rather conservative, another option could have been to take the technological performance of the best-performing installations, as in the phase III benchmarking (Ecofys et al., 2009).

Figure 4.19 shows the decomposition of the EU ETS allowances surplus over time. The EUAs surplus is the sum of the activity effect, the trade effect and the technology effect; which are positive respectively when production is lower than the production equivalent associated with the cap ($Q^{A_t}_{cement}$), when net imports are positive, and when cement carbon intensity is lower than the 2005 EU 28 level. Overallocation, the sum of the activity and trade effects, can be negative (in this case there is underallocation) when cement production is high and/or the region is exporting clinker. It can also be higher than the EUAs surplus if the technology effect is negative (high cement carbon intensity). The activity and trade effect can cancel each other out, leading to no overallocation, for example when a region is producing a high quantity of cement but importing clinker.

We also add to the EUAs surplus the offset credits used by the cement industry to show the "real" allowances surplus. Indeed, European authorities allowed companies to use offset credits (CERs or ERUs) to meet emissions caps during

[35] We use the EU ETS value of CEF_{PRO} in this section, as the heart of the question is the EUAs surplus and not "real" emissions.

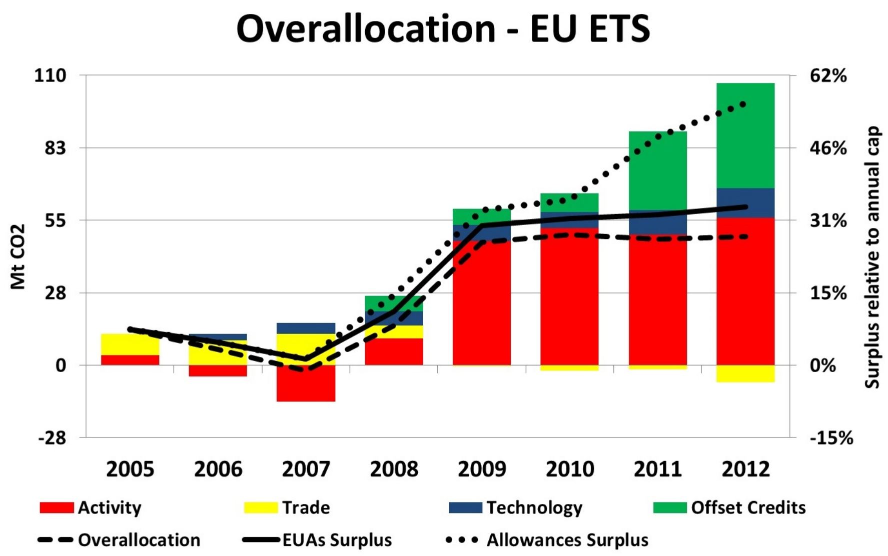


Figure 4.19: Overallocation over time in the EU ETS perimeter.

phase II. The offset limit as a share of allocations was not harmonized at the European level but differed among member states: 22% for Germany for example but only 8% in the UK (Vasa, 2012). Companies could directly finance projects and receive offset credits or purchase offset credits in the secondary market (including pure swapping of EUAs to exploit the spread and maximize trading profits).

The first year that the EU ETS came into force, the overall cap was slightly too generous with an overallocation of 12 million EUAs (roughly 8% of the cap). The increase in production in the following two years because of economic growth, and a housing bubble in certain countries, while the cap was unchanged, led to a reduction in the overallocation. Given European production levels at that time, there would have been underallocation had net imports not been so massive. In 2005, 2006 and 2007, roughly 30 million EUAs were saved thanks to the outsourcing of 9, 11 and 14 Mt of clinker respectively.

The economic downturn after 2008 led to a sharp decrease in production and therefore a massive surplus of EUAs. We estimate that the low level of activity brought 47, 52, 50 and 56 million of overallocated EUAs in 2009 to 2012 respectively (between 25% and 32% of the annual cap). After 2009, Europe became a net exporter of clinker (up to 6 Mt of clinker in 2012), so the clinker trade effect brought negative overallocation (e.g. underallocation) of 10 million EUAs (1.5% of the cap). Of the EUAs surplus for phases I and II, 45 million EUAs (3% of the cap) can be attributed to the technology effect.[36] For phase II, 84% of the surplus of 248 million EUAs was due to overallocation.

While having an excess of allowances, companies made intensive use of project-based credits. Sandbag data at the installation level reveals that virtually all cement installations used offset credits, and that the overwhelming majority of them surrendered credits up to a fixed share of allocation, which can be inferred at the maximum amount authorized for cement installations in each country:[37] 22% in Germany, 13.5% in France, 7.9% in Spain, 8.0% in the UK, 7.5% in Italy and 10% in Poland. In total, 89 million offset credits were used, representing 10% of the cap. The total surplus for phase II was then 337 million allowances, representing almost the equivalent of two years of allocation.

Figure 4.20 displays the decomposition[38] of the phase I and II allowances surplus at the EU ETS level and for the main European producers.[39] A complete year by year decomposition (as in Figure 4.19) is available in the Appendix 4.6.3 for each country.

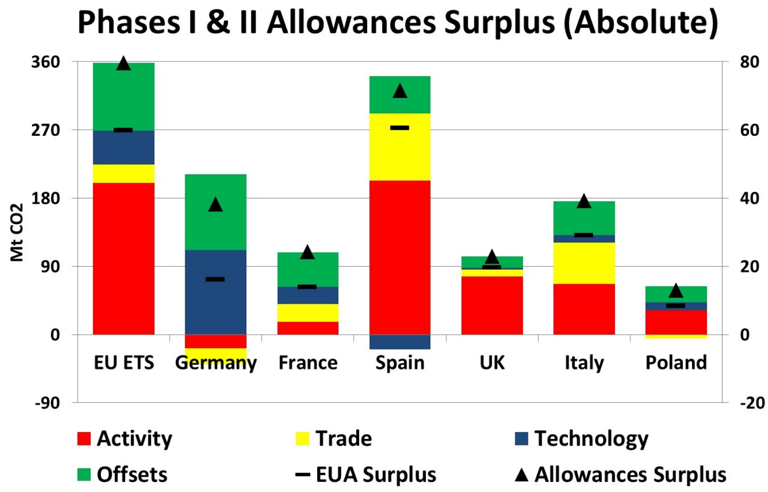
The cumulated overallocation at the EU ETS level for phases I and II is estimated at 224 million EUAs (89% due to the activity effect and 11% due to the trade effect). The country with by far the highest overallocation is Spain (65 mil-

[36]This corresponds to a "Freeze" scenario for which cement production would have been equal to the production equivalent associated with the cap, which was higher than actual production, which is why this figure is higher than the "Freeze" scenario of the previous section (42 million).

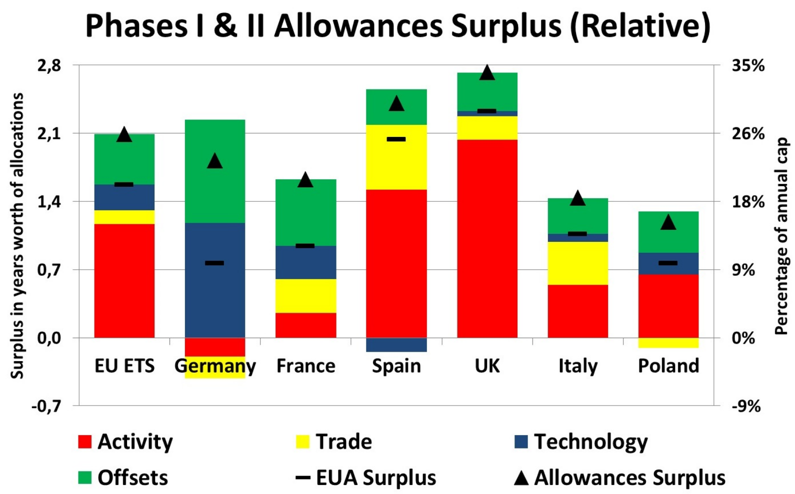
[37]For Spain (20.6%) and Italy (15%), there is a discrepancy between the share of allocation authorized at the *national* level in Vasa (2012) and the one we found at the cement installation level. An explanation could be that in these two countries the proportion of offset was probably differentiated among sectors at the installation level.

[38]For computing overallocation per country we chose to consider a European average benchmark rather than a national benchmark to put each country on an equal footing. However as the guiding principle of allocations in phase I and II was grandfathering, we also performed computations with national carbon intensities of cement (respectively 618, 637, 672, 710, 644 and 660 $kgCO_2$ per ton of cement in Germany, France, Spain, the UK, Italy and Poland). For a given cap, a lower carbon intensity of cement will correspond to a higher equivalent quantity of cement, and thus to a higher surplus and higher overallocation profits due to activity. The only noticeable difference is for Germany: because of its high technological performance, the alternative computation led to a smaller technology effect and a greater (and positive) activity effect, and thus no underallocation. For the other countries the differences are not significant.

[39]In this section, for UK at the national level we use EU ETS data rather than GNR corrected data though the coverage is incomplete, because the key question is the EUAs surplus.



(a) Decomposition of the allowances surplus (left axis for the EU ETS perimeter, right axis for main European countries)



(b) Decomposition of the allowances surplus, in years worth of allocations or relative to average annual cap

Figure 4.20: Decomposition of the allowances surplus

lion EUAs) followed by Italy (27 million EUAs), because of massive clinker imports in phase I and large falls in production in phase II.

In these two countries, while the overallocation in phase II is overwhelmingly dominated by the activity effect, the impact of the trade effect on cumulated overallocation for phases I and II is significant (33% for Spain and 42% for Italy). Indeed there was a negative activity effect in phase I (higher production than the production equivalent associated with the cap) which cancels out some of the positive activity effect (underproduction) in phase II. Conversely, there was no significant negative trade effect in phase II to cancel out the positive trade effect in phase I. Italy continued to be a clinker importer in phase II while Spain's net exports after 2009 were much smaller in magnitude than its net imports before the crisis.

Overallocation was also positive in France (9 million EUAs, more than half of it due to trade) and Poland (6 million EUAs, with a negative trade effect), while there was actually underallocation in Germany (minus 9 million EUAs, due to high production after 2008 and clinker exports). In relative terms, overallocation was also the highest in Spain and the UK (2.2 years worth of allocations) followed by Italy (0.9).

The technological performance varies significantly across countries and so does the share of the technology effect in the EUAs surplus (which is 45 million EUAs at the EU ETS level). Germany ranks first with 25 million EUAs earned thanks to low cement carbon intensity, followed by France (5 million EUAs). The technology effect is very small in the UK, Italy and Poland (0, 2, and 2 million EUAs respectively) and even negative in Spain (minus 4 million EUAs). In relative terms, Germany is also first (1.2 years worth of allocations) followed by France (0.3).

As mentioned above, because the thresholds vary from country to country, the number of surrendered offset credits varied significantly among member states. During phase II, they represented 21.1%, 13.3%, 7.4%, 6.7%, 7.2% and 8.4% of annual EUAs cap in Germany, France, Spain, the UK, Italy and Poland respectively. These differences raise concerns about equity between member states, that add to concerns about the equity of national allocation plans. Fortunately, member states with the most stringent allocation plans were generally the most generous regarding the use of offset credits (see Table 4.5). The use of offset credits has made Germany's allowances surplus more than double the EUAs surplus (38 million allowances compared to 16 million EUAs, representing an increase of 140%). The impact of offset credits on the surplus was also relatively significant in France (+72%) and Poland (+55%) but less so in Italy (+34%), Spain (+18%), and the UK (+17%).

Table 4.5: Decomposition of phase II allowances surplus

	EU ETS	Germany	France	Spain	UK	Italy	Poland
EUAs Surplus (millions)	248	7	14	58	19	34	5
EUAs Surplus (% cap)	28%	7%	18%	39%	39%	25%	9%
Technology Effect (MEUAs)	38	17	3	-3	1	1	2
Activity Effect (MEUAs)	215	-7	8	62	17	28	3
Trade Effect (MEUAs)	-5	-3	3	-1	1	6	0
Overallocation (% Surplus)	84%	-138%	77%	105%	93%	97%	54%
Offsets (millions)	89	22	10	11	3	10	5
Offsets (% cap)	10.4%	21.1%	13.7%	7.4%	8.0%	7.3%	8.5%

After having decomposed the allowances surplus and computed overallocation, let us turn to overallocation profits and offset savings. To estimate overallocation profits, we multiply yearly overallocation by the yearly average allowance spot price.[40] If negative, overallocation profits correspond to underallocation profit losses. If overallocation can be estimated with a high degree of accuracy, overallocation profits are more difficult to estimate because carbon prices vary within a year and, more importantly, allowances can be banked except from phase I to phase II. It is well-known that companies have kept a significant share of allowances as a hedge against a future scarcity).

To estimate a lower limit of savings brought by offset credits, we multiply yearly surrendered offset credits by yearly EUA-CER spread values given by Stephan et al. (2014) (4.05€, 1.54€, 2.06€, 3.34€ and 4.87€ from 2008 to 2012 respectively). Actual savings are higher for two reasons. First, if companies originate the projects, actual costs of project-based credits are much lower and savings are thus higher (for example a HFC gas project can bring offset credits less than a few Euros per ton of CO_2, bringing more than ten Euros of savings per allowance before 2012). Second, the use of offset credits increased the global cap and therefore

[40]Obtained by Tendances Carbone of CDC Climat (http://www.cdcclimat.com/-publications-.html), from 2005 to 2012 respectively: 18.04€, 17.3€, 0.7€, 22.2€, 13.1€, 14.3€, 13.0€ and 7.4€).

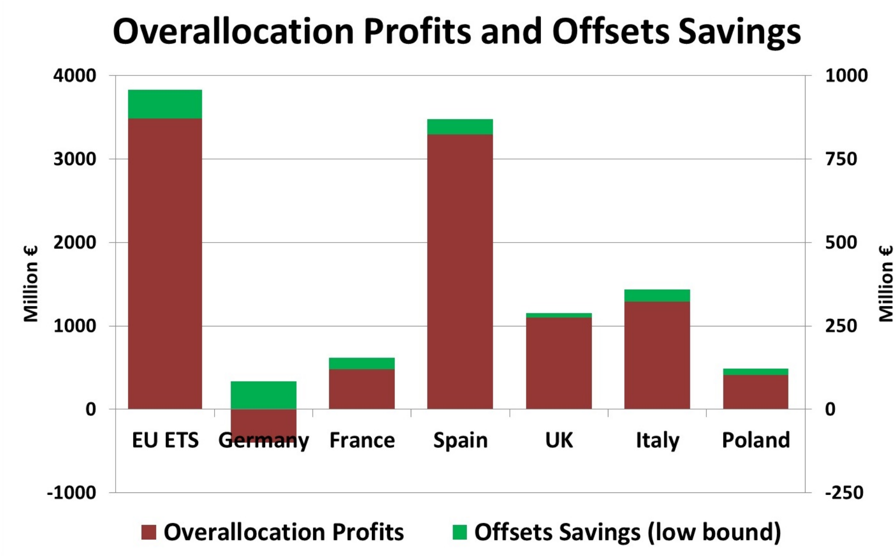


Figure 4.21: Overallocation profits and offset savings in the EU ETS perimeter (left axis) and main European countries (right axis)

decreased the EUA price (Stephan et al., 2014).

Results are shown in Figure 4.21. We estimate overallocation profits at the EU ETS level at 3.5 billion Euros. Overallocation profits would have been higher with higher EUA prices, but the latter dropped precisely because of a surplus of allowances, which was the main cause of the overallocation profits. However the EUA price would have been higher had the offset credits not been authorized. The country with by far the highest overallocation profits is Spain (824 M€) followed by Italy (324 M€) and the UK (275 M€). Then come France (120 M€) and Poland (103 M€). Germany has 100 M€ of underallocation profit losses. A low bound of offset savings is assessed at 342 M€ at the EU ETS level, and Germany is the country that benefits the most with 83 M€.

Cumulated overallocation profits for the six countries reported is around 1.5 billion Euros, i.e. slightly less than half of overallocation profits at the EU ETS level, whereas they account for two thirds of allocations. We can surmise (based on EUTL data of EUAs surplus) that overallocation profits were particularly high because of the activity effect in Romania, Bulgaria, Greece, Cyprus, Hungary and Ireland where the accumulated EUAs surplus in phase II roughly corresponded to two years of allocations.

Discussing overallocation profits by company and not by country is also relevant as the European cement market is dominated by a few multinational companies (see Table 4.6). In the case of the EU ETS, we find that the five and fifteen biggest firms account for 56% and 86% of emissions in phase II respectively. Unfortunately, the GNR database only distinguishes countries and not companies, so the only information we can obtain is the EUAs surplus through the EUTL database and the offset credits used through the Sandbag database. A rough estimate (considering that among the total 3.5 G€ of overallocation profits, companies' overallocation profits are proportional to their EUAs surplus) leads to overallocation profits of 679M€, 436M€, 370M€, 364M€ and 328M€ for Lafarge, HeidelbergCement, Holcim, Cemex and Italcementi respectively.

Table 4.6: The major European cement producers were present in many different countries in 2012

	% Emissions Phase II	Countries EU ETS	Germany	France	Spain	UK	Italy	Poland
Lafarge	15%	11	X	X	X	X		X
Heidelberg	14%	11	X			X		X
Holcim	10%	10	X	X	X		X	
Italcementi	11%	6		X	X		X	
Cemex	7%	5	X		X	X		X
Buzzi	7%	5	X				X	X

Lafarge declared in its annual reports,[41] from 2008 to 2012, 605 M€ of gains due to excess rights over actual emissions. This figure is not directly comparable to our estimation of overallocation profits, because an unknown (but small) fraction is due to technological performance (not considered as overallocation profits in our estimation), and more importantly because allowances can be banked.

Furthermore, our definition of overallocation profits leaves out offset credits. However, surrendering offset credits allows more EUAs to be banked (almost 40%

[41] Lafarge annual reports for the years year 2008 to 2012 (Lafarge, 2009, 2010, 2011, 2012) in pages F29 for 2009 to 2011 and F31 for 2012. The gains are 85 M€, 142M€, 158M€, 136M€ and 84 M€ from 2008 to 2012 respectively. The same sentence is also copied and pasted into each annual report in year X: "For year X+1, based on our estimate of allowances to be received and based on our current production forecasts, which may evolve in case of market trends different that from those expected of today, the allowances granted should exceed our needs on a consolidated basis."

more at the European level for phase II), and therefore increases gains due to excess rights over actual emissions which are reported by companies.

Based on all these points, we can estimate than the biggest European producer has sold at least half of the EUAs surplus, and we can infer a similar situation for the whole cement industry. Indeed, companies faced cash constraints because of the economic recession and selling EUAs provided an easy access to liquidity. These EUAs transfers have added to the downward trend of the carbon price (IETA, 2012), in turn decreasing the value of overallocation profits.

4.5 Conclusion

We have analysed and quantified the key drivers of carbon emissions from 1990 to 2012 in the European cement industry using an LMDI decomposition analysis. Most of the emissions changes in the EU 27 can be attributed to the activity effect. The clinker trade effect has counterbalanced approximately one third of the high activity effect in 2005-2008, because of high clinker imports at that period when production capacities were fully employed. In addition, since the 1990s there has been a slow trend of emissions reductions mostly due to the clinker share effect, but also to the fuel mix effect and the electricity emissions factor effect. Next, we have estimated technological abatement induced by the EU ETS. Because of a small acceleration in clinker reduction and alternative fuel use after 2005, 26 Mtons of CO_2 ($\pm$16 Mtons) of emissions have been abated from 2005 to 2012, corresponding to a 2.2% ($\pm$ 1.3%) decrease. However these effects could have been due to the rise of energy prices rather than the EU ETS. Finally, decomposing the allowance surplus allowed overallocation and thus overallocation profits to be assessed. The cement industry reaped 3.5 billion Euros of overallocation profits during phases I and II, mainly because of the slowdown in production, while allowance caps were unchanged.

European cement companies have been suffering from the economic downturn through reduced sales, low return on investment (BCG, 2013) and a decline in profits (Bolscher et al., 2013). However, their financial situation would have been far worse had the EU ETS not been implemented. During phase II, the scheme was tantamount to a subsidy of 3.5 Euros per tonne of cement produced in Europe,[42] which significantly increased the profitability of the sector.[43] Presented as a threat to competitiveness, the EU ETS has paradoxically boosted European cement industry competitiveness, when defined as a company's ability to earn (Quirion, 2010).

Since 2013 and the beginning of phase III, the EU ETS conditions have been less favorable for the cement industry, because of an increased stringency of the allocation methodology. The allocation is now based on the average of the 10% best-performing installations, corresponding to 766 kgCO_2 per ton of clinker (European Commission, 2011), and declines at a rate of 1.74% per year. However, this benchmark is then multiplied for each installation by the historical activity level (HAL), which is generally based on pre-crisis levels.[44] Overallocation occurred in 2013 (emissions were 20% lower than the cap) and will go on for the years to come unless production significantly increases.

Because of high levels of uncertainty concerning future production levels, the difference between HAL and actual production can be very large. The choice of HAL thus has deep financial repercussions on companies: too high a HAL automatically brings overallocation profits while one that is too low induces losses of profit. Output-based allocations, which consists in linking directly allocations to production, have the desirable benefit of by-passing the determination of HAL and the potential overallocation profits or underallocation profit losses that accompany it. Given the order of magnitude of financial values at stake that have been reported in this analysis, this advantage outweighs by far potential drawbacks.[45] Such a dynamic allocation (**?**) would lead to fewer economic distor-

[42] 3.5 billion Euros of overallocation profits divided by 1 billion tonnes of cement produced.

[43] Based on financial data (including reported sales of allowances), Boyer and Ponssard (2013) find that the EBIDTA/sales ratio (Earnings Before Investment, Depreciation, Taxes and Amortization) of the Western European cement industry for phase II would have been 26.3% without sales of allowances, instead of 32.9%. Furthermore, the impact would have been more significant had cement companies sold all these financial assets instead of banking a significant share.

[44] The HAL is, except for changes in capacity, the median value of the annual activity during the period 2005-2008 or 2009-2010 (whichever is the highest) (European Commission, 2011).

[45] These drawbacks include fluctuating cap, administrative complexity, or providing little incentive to reduce the consumption of polluting goods (Quirion, 2009). However, these defaults can be mitigated with an appropriate design and additional policies. An extended discussion of optimal design for output-based allocations and complementary policies, as well as the comparison between other anti-leakage policies is beyond the scope of this chapter. For more information we recommend Chapter 1, Dröge (2009), Meunier et al. (2014) and Neuhoff et al. (2014).

tions and more incentives to reduce carbon emissions than the current allocation methodology. An alternative option would be full auctioning with the inclusion of importers (Neuhoff et al., 2014).

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4.6 Appendix

4.6.1 Formulas

$$\triangle^{act-F} = \frac{C_{F,T} - C_{F,0}}{\ln(C_{F,T}) - \ln(C_{F,0})} \ln(\frac{Q_{cement,T}}{Q_{cement,0}}) \tag{4.15}$$

$$\triangle^{sha-F} = \frac{C_{F,T} - C_{F,0}}{\ln(C_{F,T}) - \ln(C_{F,0})} \ln(\frac{R_T}{R_0}) \tag{4.16}$$

$$\triangle^{tra-F} = \frac{C_{F,T} - C_{F,0}}{\ln(C_{F,T}) - \ln(C_{F,0})} \ln(\frac{H_T}{H_0}) \tag{4.17}$$

$$\triangle^{fmix} = \frac{C_{F,T} - C_{F,0}}{\ln(C_{F,T}) - \ln(C_{F,0})} \ln(\frac{CEF_{F,T}}{CEF_{F,0}}) \tag{4.18}$$

$$\triangle^{eff-F} = \frac{C_{F,T} - C_{F,0}}{\ln(C_{F,T}) - \ln(C_{F,0})} \ln(\frac{I_{T,T}}{I_{T,0}}) \tag{4.19}$$

$$\triangle^{act-P} = \frac{C_{P,T} - C_{P,0}}{\ln(C_{P,T}) - \ln(C_{P,0})} \ln(\frac{Q_{cement,T}}{Q_{cement,0}}) \tag{4.20}$$

$$\triangle^{sha-P} = \frac{C_{P,T} - C_{P,0}}{\ln(C_{P,T}) - \ln(C_{P,0})} \ln(\frac{R_T}{R_0}) \tag{4.21}$$

$$\triangle^{tra-P} = \frac{C_{P,T} - C_{P,0}}{\ln(C_{P,T}) - \ln(C_{P,0})} \ln(\frac{H_T}{H_0}) \tag{4.22}$$

$$\triangle^{act-E} = \frac{C_{E,T} - C_{E,0}}{\ln(C_{E,T}) - \ln(C_{E,0})} \ln(\frac{Q_{cement,T}}{Q_{cement,0}}) \tag{4.23}$$

$$\triangle^{eff-E} = \frac{C_{E,T} - C_{E,0}}{\ln(C_{E,T}) - \ln(C_{E,0})} \ln(\frac{I_{El,T}}{I_{El,0}}) \tag{4.24}$$

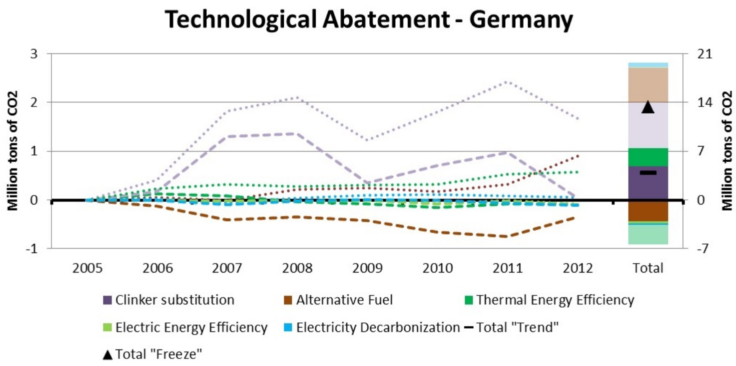


Figure 4.22: Technological abatement in Germany. The curves on the left side show the abatement due to the different effects under the "Freeze" scenario (dotted line) and the "Trend" scenario (dashed line). The histogram on the right gives the sum of abatements over the years, in full color for the "Trend" scenario, and in full color *plus* faded color for the "Freeze" scenario.

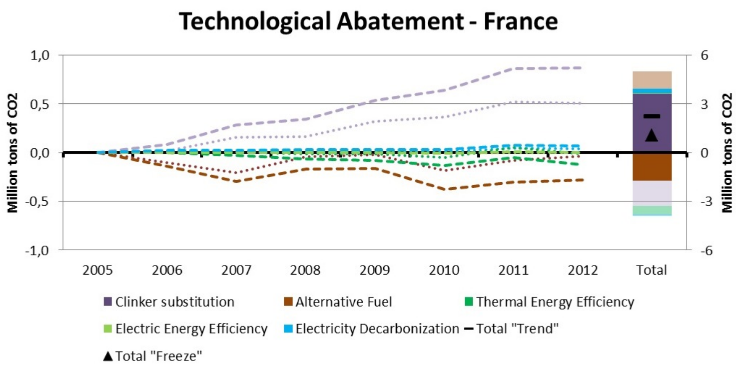


Figure 4.23: Technological abatement in France.

$$\triangle^{Celec} = \frac{C_{E,T} - C_{E,0}}{\ln(C_{E,T}) - \ln(C_{E,0})} \ln\left(\frac{CEF_{elec,T}}{CEF_{elec,0}}\right) \tag{4.25}$$

4.6.2 Technological abatement country by country

4.6.3 Overallocation country by country

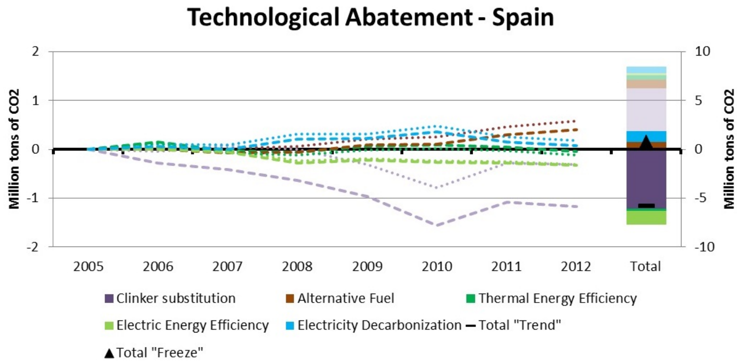


Figure 4.24: Technological abatement in Spain.

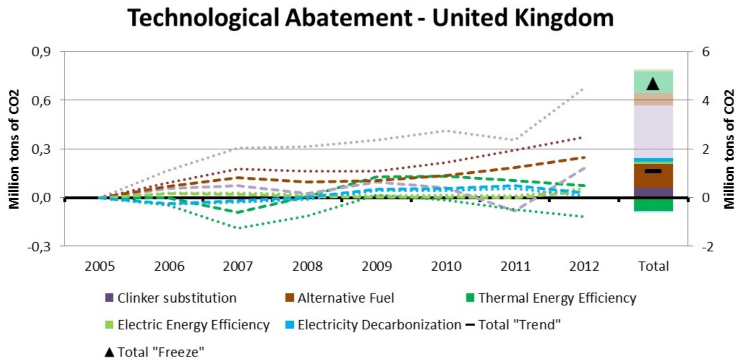


Figure 4.25: Technological abatement in the UK.

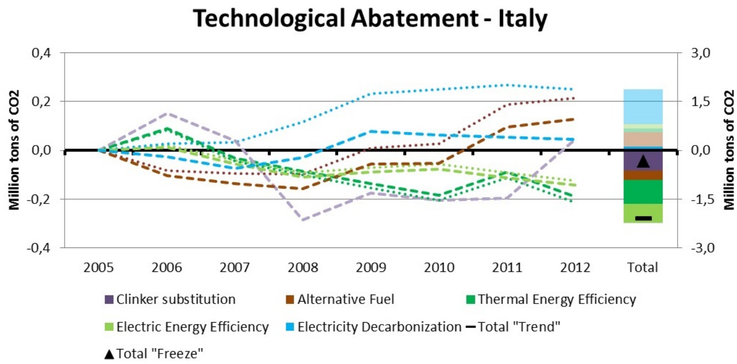


Figure 4.26: Technological abatement in Italy.

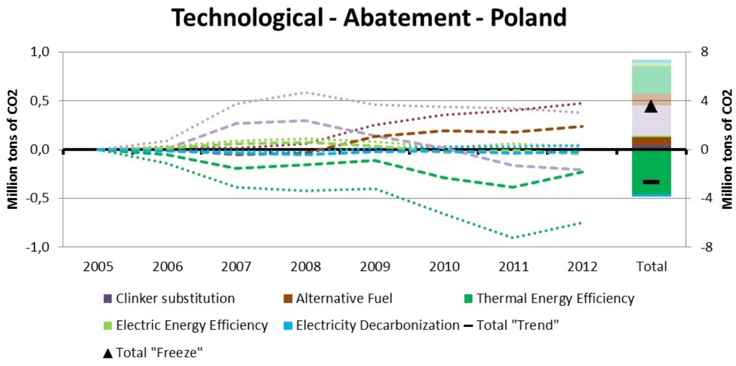


Figure 4.27: Technological abatement in Poland.

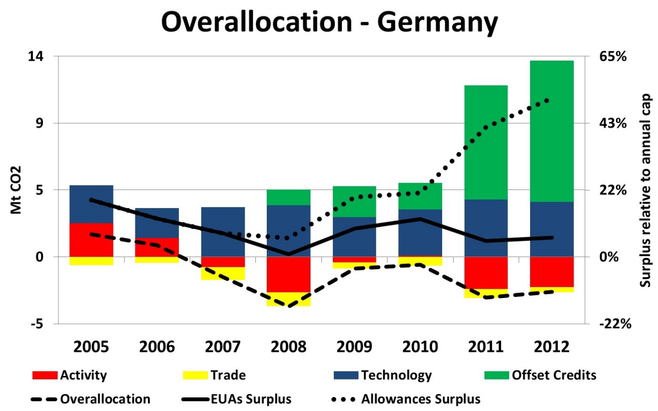


Figure 4.28: Overallocation in Germany.

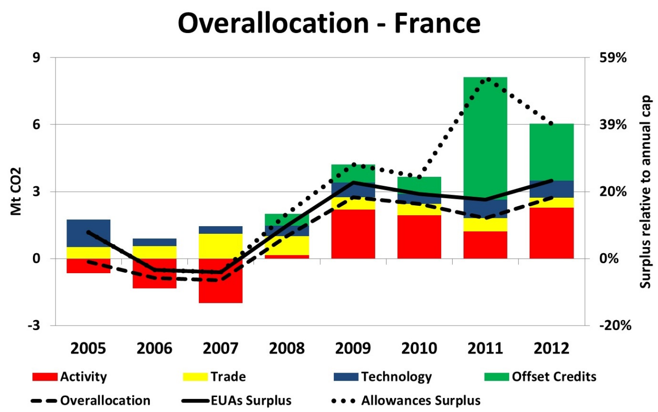


Figure 4.29: Overallocation in France.

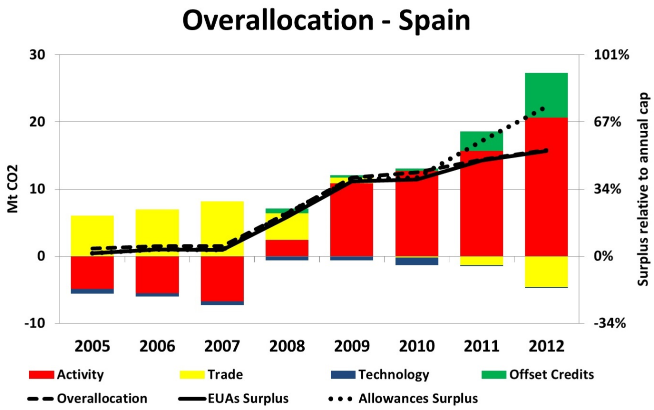


Figure 4.30: Overallocation in Spain

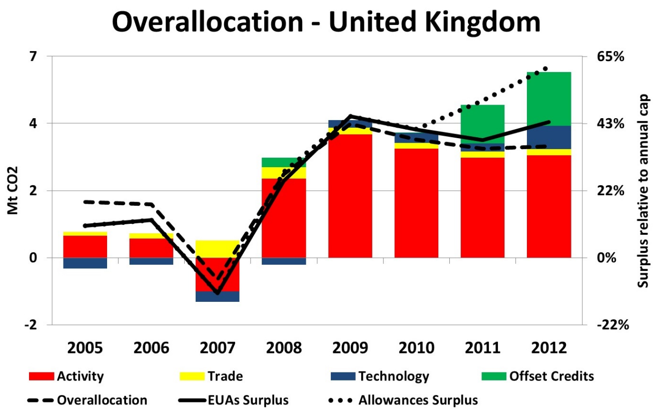


Figure 4.31: Overallocation in the UK

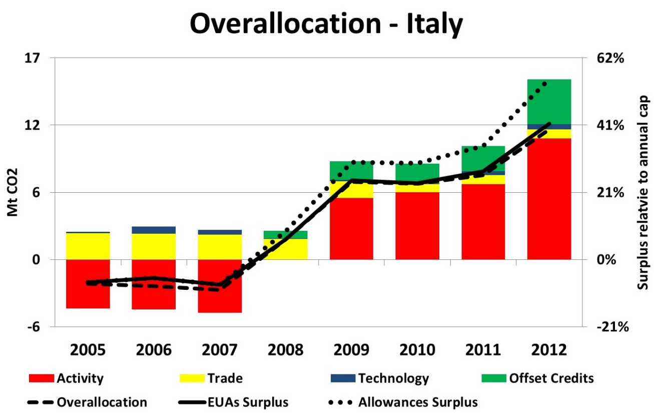


Figure 4.32: Overallocation in Italy

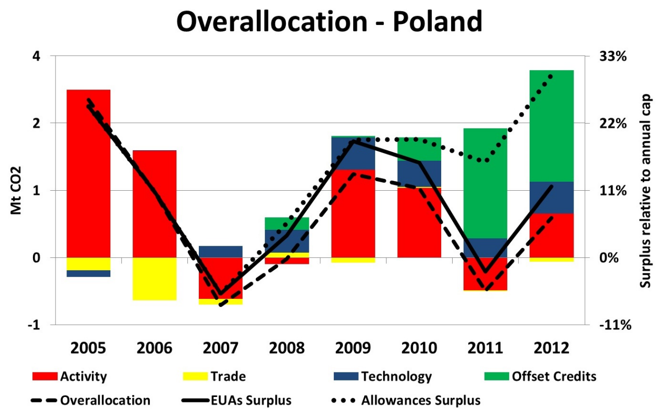


Figure 4.33: Overallocation in Poland

5

EU ETS, Free Allocations and Activity Level Thresholds: The devil lies in the details

Starting from Phase 3, the EU Emissions Trading System introduced a new rule which links the level of free allocation to the activity level of an installation – known as activity level thresholds (ALTs). Whilst put in place with the intention to reduce excess free allocation to low-activity plants, the new rule creates incentives for installations to "game" output levels in order to maximise free allocation. This chapter measures the distortionary effects resulting from ALTs, by exploiting the natural experiment of the introduction of the new rule in 2012, and discusses whether the disadvantages of ALTs outweigh the advantages.

The justification for using free allocations in emission trading schemes has evolved over time. Historically, in schemes such as the U.S. acid rain program, it was introduced as a compensation mechanism for the owners of existing industrial assets for a change in the rules of the game (Ellerman et al., 2000). A lump sum transfer would be made to existing assets through a predetermined amount of annual free allocations for a given number of years. Such methods are termed "grandfathering", "historic", "lump-sum" or "*ex-ante*" allocation. New assets would not be allowed free allocations and thus would have to pay for all their permits on the market. As long as the free allocations are predetermined, all assets (old and new) would compete on the same playing field, the price of permits would provide the same opportunity cost for mitigating pollution, and in theory, the output price of the goods sold would incorporate the price signal for consumers.

More recently, free allocations have been explicitly used (or have been proposed to be used) as a way to strategically alleviate the risk of offshoring production and emissions (so-called "carbon leakage") for Energy-Intensive and Trade-Exposed (EITE) sectors such as cement, chemicals and steel. Economists generally agree that, in a world of unequal carbon prices, full auctioning together with some form of border levelling of prices would be the second best approach to tackling leakage (Hepburn et al., 2006; Monjon and Quirion, 2011). However, the required degree of international cooperation to achieve such a system has not yet been forthcoming. Thus, a number of papers suggest that, from an economic efficiency standpoint, "output-based" allocation (OBA) would be a preferred third-best option (Fischer and Fox, 2007; Quirion, 2009; Fischer and Fox, 2012; Meunier et al., 2014b). Under OBA, the volume of free allocation is directly proportional to actual production, hence it acts as an implicit production subsidy and thus provides little incentive to reflect the carbon price in the final product price, or to reduce production of the polluting goods. The output reduction is then lower than the social optimum, inducing higher overall costs for a given emissions reductions target. However, these cost inefficiencies may be balanced out by the reduction in windfall profits and carbon leakage compared to grandfathering, if applied to targeted sectors. An OBA scheme has been implemented within the Californian ETS which began in 2012 (California Air Resources Board, 2013). In contrast the EU ETS Phase 3 is unique in using a complex system. It combines an *ex-ante* calculation[1] of an allocation and subsequent lump-sum transfer based on historic

[1] Note that *ex-ante* and *ex-post* refer to whether the calculation of the freely allocated amount of allowances occurs prior to or following the production and emissions for which allowances are

output (and multiplied by an emissions intensity benchmark) with a possible *ex-post* calculation and adjustment of this lump-sum according to rules related to actual capacity and activity levels as defined in Decision (2011/278/EU) (European Commission, 2011). Situations in which *ex-post* adjustments occur include the arrival of new entrants into the market, plant capacity extension/reduction, plant closure and partial cessation or recommencement of activity at an existing plant. These latter rules are governed by the activity level thresholds (ALTs).[2]

Qualitatively, ETS schemes with ALTs approximate OBA: the amount of free allocations will vary with the activity level, and the over allocation profits[3] associated with *ex-ante* schemes will be reduced.[4] The advantage of ALTs rules is that they allow for a fixed cap (in fact a cap which will not exceed a predetermined amount for existing installations and the reserve for new entrants). One disadvantage is that they introduce an element of complexity in the scheme. Under these non-linear rules, the lump sum transfer of allowances to EITE sectors is reduced by 50%, 75% or 100% if the annual level of production of the plant falls below 50%, 25% or 10% respectively, of the historical activity level (HAL) of production that is used to determine the *ex-ante* allocation (European Commission, 2011).

A second disadvantage is that the ALTs introduce distortions, which is the focus of this chapter. A recent study on the EU ETS impacts on the cement sector 2005-2013 (Neuhoff et al., 2014) found preliminary evidence through data analysis and comprehensive interviews with industry executives, that new ALTs introduced in 2013 provided cement installations the incentive to adjust output levels. The rationale is as follows. Since the free allocation in year $t + 1$ is directly linked to output in year t, if output levels lie below the threshold levels, there may be an

to be allocated.

[2]New entrant provision and closing rules were already in place in Phases 1 and 2 of the EU-ETS but they differed among Member States. A closure rule is also used in the Californian ETS.

[3]Overallocation profits come from the allowances surplus automatically generated when the number of free allowances received is higher than emissions necessary to manufacture the amount of cement produced (Chapter 4). Overallocation profits can be distinguished from windfall profits, which refer to the profits from free allocation where emitters additionally profit from passing on the marginal CO_2 opportunity cost to product prices, despite receiving the allowances for free. overallocation profits can occur even in the absence of cost pass through, if output fall short of historic levels.

[4]Windfall and overallocation gains have been a persistent shortcoming of the use of *ex-ante* free-allocation mechanism in the EU ETS (*e.g.* Laing et al. (2014); Sartor et al. (2014); Sandbag (2012).

incentive to increase output in year t to achieve the relevant threshold (.10, .25, .50) and receive higher free allocations in year $t+1$. In this chapter, such strategic adjustments of output motivated by ALTs is termed "gaming" behaviour, in line with the management literature (*e.g.* Jensen (2001)). Neuhoff et al. (2014) report that in interviews, company executives consistently confirm these practices where the regional cement market demand is insufficient to reach the minimum activity level. They identify three channels to marginally increase production in a plant which is producing below the threshold:

- Production shifting among local plants, *i.e.* reducing the production at a plant which is well above the threshold to increase the production at the plant which is below; this generates some transport costs[5] so that it can be too costly to be undertaken at a large scale;

- Exports of clinker to other markets so as not to perturb the local market while increasing production; this generates some cost in terms of export price rebate, since these exports would not naturally occur;

- Increase the clinker to cement ratio, *i.e.* incorporate within limits more clinker in cement instead of using less costly cementitious additives such as slag or flying ashes; this directly generates some cost.

In this chapter we revisit the existence and the magnitude of the distortions, and ask whether or not the installation outputs and trade flows in 2012 affected by the free allocation policy change for year 2013. Our analysis is conducted in a unique context of low demand induced by a severe economic downturn. The construction of a counterfactual requires some assumptions, the most significant of which considers that consumption and price levels for cement are independent of the allocation scheme. This assumption is consistent with the observations made in Neuhoff et al. (2014). We discuss in detail how our results would be affected if we had adopted the more standard assumption in which grandfathering and output based allocation would lead to different cement and price levels.

[5] McKinsey&Company (2008) estimate that transport costs for a tonne of clinker from Alexandria to Rotterdam are roughly €20/tonne, and that inland shipping costs are approximately €3.5/tonne per 100km and inland road transport was about €8.6/ton per 100km.

Empirical studies on the impact of ALT or similar rules remain limited. Most of these studies have examined the distortive effects of combined *ex-ante* allocations with *ex-post* new entrant and plant closure provisions. Pahle et al. (2011); Ellerman (2008); Neuhoff et al. (2006) compared the new entrant provision relative to auctioning. These papers argued that new entrant provisions distort via their impact on investment decisions in the electricity sector (essentially by acting as a subsidy). Meunier et al. (2014b) compared this same provision with an output-based scheme whenever firms face an uncertain demand in the EU cement sector. They showed the entrant provision could induce excessive new investments while offering limited protection against leakage. Fowlie et al. (2012), this time for the US cement sector, compare *ex-ante* schemes with closure rules with an output-based scheme and show that the lifetime of old inefficient plants would be unduly extended with the former while temporarily reducing leakage. Only this last paper has discussed the impacts of the possible distortions associated with the (limited) addition of "non-linear" *ex-post* adjustments to *ex-ante* allocation via the use of ALTs, such as introduced in the EU ETS Phase 3 (2013-2020).

The findings in this chapter could be potentially relevant to other EITEs with similar characteristics. Altogether, we argue that the benefits of implementing ALTs in terms of reduced overallocation profits will not necessarily outweigh the significant costs in the form of distortions. Hence it may be preferable to abandon ALTs for OBA for some sectors in the short run. We discuss some broader questions if such a change were adopted.

The chapter is organized as follows. Section 5.1 discusses the EU ETS Phase 3 allocation rules, the predicted gaming behaviour from thresholds and the alternative allocation rules. Section 5.2 describes our conceptual framework for evaluating the effects of ALTs, the methodology, data sources, as well as the key assumptions involved in our analysis. Section 5.3 presents the results. Section 5.4 concludes and discusses policy recommendations.

5.1 ETS free allocation rules and gaming of ALTs

5.1.1 The EU-ETS Phase 3 free allocation rules

In Phase 3 of the EU ETS, installations in sectors "deemed to be exposed to carbon leakage" are eligible to receive free allocation of emission allowances. The determination of the free allowances for each installation combines an *ex-ante* calculation, based on the historic output for existing installations (known as the "historical activity level" or "HAL"[6]) or the initial capacity for new installations, with an *ex-post* calculation based on the ongoing activity level of this installation as defined in Decision (2011/278/EU) (European Commission, 2011). The *ex-post* calculation provides step wise adjustments intended to reflect changes in market volumes. These adjustments follow complex procedures.

For existing installations, the precise relationship that determines the next-period allocation from *ex-ante* and *ex-post* values is summarised by Equations (5.1) and (5.2) below. The amount of free allocations to an installation, i, at period $t+1$, for an eligible product, p is denoted $A_{i,p,t+1}$.

$$A_{i,p,t+1} = CSCF_t \times B_p \times HAL_{i,p} \times ALCF_{t+1}\left(\frac{q_{i,p,t}}{HAL_{i,p}}\right) \tag{5.1}$$

In equation (5.1) $CSCF_t$ is the uniform cross-sectoral correction factor,[7] B_p is the benchmark for product p,[8] $HAL_{i,p}$ represents the historical activity level, $q_{i,p,t}$ represents the output of the eligible product in year t; and $ALCF_{t+1}(\frac{q_{i,p,t}}{HAL_{i,p}})$ is the activity level correction factor. The latter factor defines a step wise function

[6]The benchmarked product-related historical activity level (HAL) is defined as maximum of the median annual historical production of the product in the installation (or sub-installation) concerned during either 2005-2008 or 2009-2010. (cf. Decision (2011/278/EU)).

[7]This is determined by comparing the sum of preliminary total annual amounts of emission allowances allocated free to installations (not electricity) for each year over the period 2013-2020. In 2013 the CSCF is equal to 0.9427, then declines at 1.74% per year.

[8]Product benchmarks in general reflect the average performance of the 10% most efficient installations in the sector or subsector in the years 2007-2008.

for the thresholds. It is defined as:

$$ALCF_{t+1}(\frac{q_t}{HAL}) = \begin{cases} 1, & \text{if } \frac{q_t}{HAL} \geq 0.5 \\ 0.5, & \text{if } 0.25 \leq \frac{q_t}{HAL} < 0.5 \\ 0.25, & \text{if } 0.10 \leq \frac{q_t}{HAL} < 0.25 \\ 0, & \text{if } 0 \leq \frac{q_t}{HAL} < 0.10 \end{cases} \tag{5.2}$$

For new installations, the historic activity level is replaced by the capacity, to be precisely determined according to the rules.[9]

5.1.2 Gaming and thresholds

Gaming behaviour refers to artificially increasing production to attain thresholds, in order to obtain more allowances. Consider a plant for which the "business as usual" activity level for year 2012 would be at say 40% of its historic activity level. Increasing production up to 50% of its historic activity level allows doubling the free allocation received. A rough calculation with a clinker plant illustrates the potential benefit of gaming. Suppose HAL refers to 1 Mt/year (millions of metric tons per year), the business as usual is 0.4 Mt in 2012 so that the plant needs to increase production by 0.1 Mt to achieve the 50% threshold. At 7.95 €/tCO_2 in 2013 (average future price of December 2013 during year 2012), if the firm gets 100% of free allowances relative to HAL it is worth 5.8 M€(0.9427×1Mt $\times$ 0.766 tCO_2/t $\times$ 7.95€/tCO_2, numbers being respectively CSCF, HAL, clinker benchmark and carbon price); losing 50% allowances implies a loss of 2.9 M€. Suppose the emission intensity is say 0.8 tCO_2/t of clinker. The increase in emissions is then equal to 0.080 tCO_2 which at 7.95 €/tCO_2 amounts to 0.64 M€.

In the presence of activity level thresholds, the allowances net benefit of gaming is the difference between the increased free allocations and the certificates needed to cover the increased production (in our case 2.26M€=2.9M€-0.64M€). The net benefit depends on the price of CO_2, the benefit rising with the price. However, this artificial increase of production involves cost inefficiencies, which can be assumed increasing function of the extra production, independent of the CO_2

[9] Guidance document number 7 in European Commission, 2011.

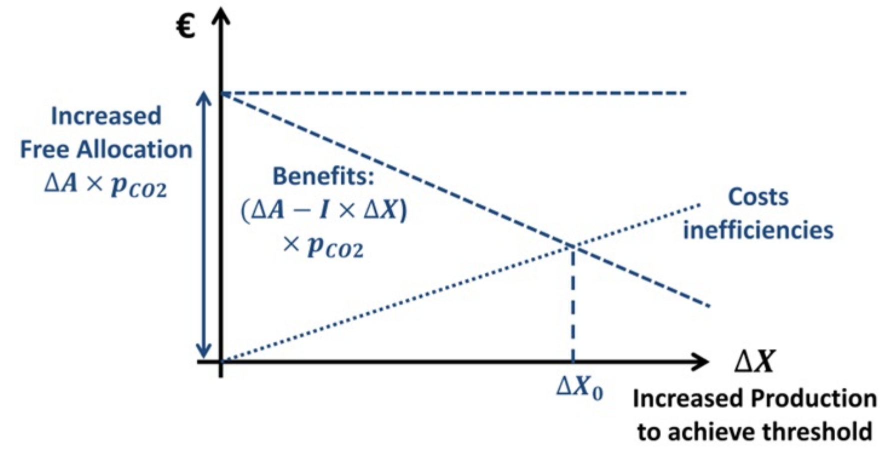


Figure 5.1: The value of gaming. The installation engages in gaming when $\Delta X < \Delta X_0$. I refers to the carbon intensity of the plant. Benefits are increased free allocations minus extra emissions.

price but dependent on the plant. These cost inefficiencies can up to a point cancel out the gains from increased free allocation. This is shown in Figure 5.1, where gaming is undertaken only if the increased production to attain the threshold is less than ΔX_0. In our case, if the extra production of 0.1 ton of clinker does not involve cost inefficiencies of more than 2.53M€, gaming is profitable.

Evidence of strong responses to thresholds – where small changes in behaviours lead to large changes in outcomes – has been found in the recent literature. Sallee and Slemrod (2012) find evidence that the automakers respond to notches in the Gas Guzzler tax and to mandatory fuel economy labels by manipulating fuel economy ratings in order to qualify for more favourable treatment. The management control literature also finds that managers tend to react strongly to the existence of a threshold. This is the case, for example, when bonuses depend on the achievement of a given level of sales for a sales manager, a given productivity indicator for a plant manager, a given return on investment for a business manager, a given level of the total shareholder return for a CEO, etc (Locke, 2001). In a well-known article, Jensen (2001) points out that such "gaming" behaviour is perfectly rational under threshold rules. He argues that these rules imply an agency cost which is largely underestimated and suggests that linear bonus schemes should be preferable.

5.1.3 Alternative free allocation rules

The EU ETS Phase 3 rules can be compared with an ex ante allocation without ALTs or an output-based allocation scheme. Under OBA, the next period allocation is determined according to an equation similar to equation (5.1) (with $q_{i,p,t} \longleftrightarrow HAL_{i,p} \times ALCF_{t+1}(\frac{q_{i,p,t}}{HAL_{i,p}}))$. The scheme therefore has no thresholds, and the historic activity level HAL is replaced by the previous year activity level $q_{i,p,t}$ so as allocations are altered on a continuous yearly production basis. In this chapter, we will evaluate the impact of the ALTs by contrasting four scenarios, with their respective acronym:

- *Ex-ante* free allocation with ALTs (Phase 3 allocation rules) and gaming (EXALTG);
- *Ex-ante* free allocation with ALTs (Phase 3 allocation rules) without gaming (EXALTNG);
- *Ex-ante* free allocation without ALTs (EX);
- *Ex-post* output based allocation (OBA).

Scenario EXALTG corresponds to what was observed in Phase 3. Scenario EXALTNG applies the same rules but it is a hypothetical scenario where no gaming behaviour is observed (every variable is identical as in EX, except the allocation, which follows a different rule). EXALTNG, EX and OBA represent counterfactuals.

5.2 Methodology and data

Since 2013 is the first year the threshold rule is in place, the 2012 activity level directly determines the allocation allowances for 2013. The preliminary analysis in Neuhoff et al. (2014) provided evidence of distortions arising from the ALTs rule. The present study quantifies these distortions.

5.2.1 The cement sector

Our analysis focuses on the cement sector[10] for three reasons. First, it ranks amongst the highest in terms of carbon intensity per value added thus the effects of free allocation rules are magnified. The cement production process can be divided into two basic stages: production of clinker and the subsequent grinding and blending of clinker with other mineral components to produce cement. The first stage (clinker production) accounts for the bulk of carbon emissions in cement production. Allocation under the EU ETS is based on a benchmark on clinker. The relevant output involved in the threshold rule is then the quantity of clinker produced. However cement is the final product and is traded as well, so that the analysis has to be done simultaneously for both products. Industry characteristics (economies of scale, sunk cost, high land transportation cost) suggest that the relevant market be defined at the regional level, which we define as EU member states.[11]

Second, as the sector experienced a demand collapse in the order of 50% or more between 2007 and 2012 in several member states, the ALTs rules were likely to have been a relevant factor for operational decisions during the period studied. Indeed we suspect that the most important differences between scenarios EX and EXALTG will occur in countries in which cement and clinker consumption in 2012 fell well short of historical consumption level and hence ALTs rules are relevant. For convenience our results obtained for each member state will be aggregated. The 26 EU ETS member states[12] with ETS-participating clinker production plants will be divided into two groups (see Table 5.1). The first group includes countries where the average domestic cement consumption in 2011-2012 was less than 70% of 2007 levels.[13] We name this group "low demand" (LD) countries. Of the LD countries, we present some of the results for Greece and Spain, as these two member states were particularly affected by the downfall. The LD coun-

[10]For an overview of the European cement sector see Boyer and Ponssard (2013).

[11]Some small countries are regrouped into larger entities which are coherent in terms of regional market (see Section C.1).

[12]Note that Iceland, Liechtenstein, Malta have no listed clinker plants in the EUTL database, while data for Cypriote plants was not able to be exploited due to missing data.

[13]The average of 2011 and 2012 was taken since both years are relevant to the analysis that follows here. 2007 is taken as the reference year since this was the year in which demand peaked in most EU Member States prior to the economic crisis of 2008.

Low Demand (LD) Countries	Moderate Demand (MD) Countries
Ireland, Spain, Greece, Bulgaria, Hungary, Denmark, Portugal, Italy, Slovenia and Baltic countries	Austria, Belgium, Czech Republic, Finland, France, Germany, Netherlands, Norway, Poland, Romania, Slovakia, Sweden and United Kingdom

Table 5.1: Moderate (MD) and low demand (LD) countries in terms of cement consumption in 2012 relative to 2007 levels

tries represented 51% of EU ETS cement emissions in 2008 and 40% in 2012. The remaining countries are classified as "moderate demand" (MD).

Third, the cement sector is characterised by relatively homogeneous products and production processes, unlike chemicals and steel for example with many product categories and differentiated impacts. This aspect does not make distortions due to ALTs more likely to occur; but facilitate their quantifications. Indeed, allocation is determined with activity levels ($\frac{q}{HAL}$, in the cement sector, q being the quantity of clinker), but data on output is not publicly available at the installation level. However, data on emissions is, thanks to the European Union Transactions Log (EUTL). Because of the very strong and direct relationship between production of clinker, a highly homogeneous product, and emissions, it is possible to infer *production* (activity) from *emissions*.[14]

5.2.2 Conceptual framework and main assumptions

The quantification of distortions due to the thresholds necessitates the elaboration of a counterfactual scenario for 2012 (what would have happened had the threshold rule not been implemented) for each relevant market. A simple compar-

[14] We use the observed ratio of publically-reported verified emissions (E) relative to the Historical Emissions Level (HEL), to proxy the share of unobserved activity level relative to Historical Activity Level (HAL) *i.e.* $E/HEL \approx q/HAL$. This approximation is possible because the emissions intensities of clinker production have changed only very marginally in the EU in recent years between 2005 and 2012 (WBCSD, 2009). At first sight, the approximation $E/HEL \approx q/HAL$ may turn problematic for precisely distinguishing between installations that are above or below thresholds (25% and 50% of q/HAL). However, as detailed in Appendix 5.5.1, we ensure that installations are correctly identified using 2013 allocations data. This reveals whether or not the installation had seen its allocation reduced because of 2012 activity levels. Further, 2013 allocation data also allowed us to obtain clinker carbon intensity at the plant level, and then to assess production through emissions (see Appendix 5.5.1).

ison between 2011 and 2012 would give inaccurate results because of underlying market trends *e.g.* cement consumption fell by 13% at the EU level between 2011 and 2012. Comparing with a counterfactual enables us to understand the magnitude of the excess output due to ALTs, and the corresponding excess emissions and overallocation profits. A straightforward caveat is that our results are then very dependent on the counterfactual, which is developed by combining historical data at the country and plant level using a panel data model. We also conduct Monte Carlo analysis to assess confidence intervals and conduct a number of robustness tests to limit this caveat.

Data at the plant level (246 clinker plants operating in 2010, 2011 and 2012[15]) from the EUTL are used to obtain the distribution of plant activity level for 2011 and 2012 EXALTG. To construct the value of counterfactual plant activity level for the other 2012 scenarios, we suppose that cement consumption and price are independent of the allocation method. We discuss this assumption in two steps.

The first step assumes that EX and OBA give identical cement consumption and price (H_1). This appears at odds with the economic literature (Fischer and Fox, 2007; Demailly and Quirion, 2006) which would clearly distinguish between *ex-ante* free allocations and *ex-post* OBA. *Ex-ante* free allocations would ordinarily not provide any protection against leakage (they are a lump sum transfer and firms marginal cost fully support the cost of carbon) while *ex-post* OBA allocations would (with OBA, firms receive free allocation proportional to their output the marginal cost is unchanged and there are no competitive impacts with respect to imports; this is the usual argument in favour of OBA). Cement consumption and price would then differ depending on which of these two allocation schemes is used: cement price would be higher in EX than OBA as firms would pass through some of the opportunity cost of allowances, and then consumption lower through a price effect. This chapter departs from this view. Rather, it assumes that firms adopt exactly the same pricing and production decisions in their home market in OBA and *ex-ante* allocation.

This assumption is supported by a series of in depth interviews with cement

[15]For this purpose, we rely heavily on the work carried out in Chapter 4, which have developed an installation level dataset for the EU cement sector with clinker producing installations identified.

sector actors (Neuhoff et al. (2014) p.26) to explicitly show why there has been no price change. These interviews point out three reasons for such behaviour: (i) The *ex-ante* free allocations have been obtained precisely to mitigate leakage thus a risk of losing future free allocations if regulators observe the ability to pass on the cost of carbon without observing leakage; (ii) long term strategic considerations – such as maintaining market share and good client relationships – could partially balance the incentive to pass the carbon price; (iii) the risk of drawing the attention of competition authorities due to abnormal profit levels if the carbon cost was indeed passed through.[16] However it is important to note that these empirical observations have been made in a context of low carbon price. We certainly do not claim that H_1 would prevail at all times.

The second step assumes that EX (or EXALTNG since these only differ in terms of allowances) and EXALTG give identical cement consumption and price (H_2). This means that cement consumption and price would not be affected by gaming. Since the clinker production is likely to increase through gaming the question is what happens to the excess production. Neuhoff et al. (2014) identify three channels: reshuffling of production among plants (this may be quite easily done since many cement companies are multi-plants), exports to non EU countries and increase in the clinker to cement ratio. From an economic point of view one needs to rationalize why a player would use these channels rather than simply pour its excess production directly into its regional market. Our answer comes from the oligopolistic nature of competition and the low price elasticity in the cement market.[17] Increasing the regional supply would most certainly depress the price substantially, and induce strong reactions from competitors. While we cannot exclude that a small fraction of the excess production does actually go into the regional market, the data will by and large support the extensive use of the other three channels.

These two hypotheses H_1 and H_2 suppose that cement consumption and price are independent of the allocation method, and allow us to construct a counterfactual plant activity common to the counterfactual scenarios (EX, EXALTNG

[16] The EU cement industry faced and continue to face investigations from EU and national competition authorities; see for instance `https://www.gov.uk/cma-cases/aggregates-cement-and-ready-mix-concrete-market-investigation`

[17] For estimates see Meunier et al. (2014a).

and OBA) in the absence of data or models to directly assess the effects of allocation methodologies on consumption and prices. We argue the empirical evidence reported in Neuhoff et al. (2014) is persuasive and support these assumptions. However, given the discrepancy with the literature, it is important to see how our results would stand if H_1 or H_2 were relaxed. This is done in Section 5.3.7, where we show that results remain mostly unchanged (especially relaxing the more controversial assumption H_1). Moreover, qualitative assessment suggests that our estimations would be biased in the conservative direction (underestimating the effect of gaming on production and profits).

We now proceed on our methodology. Having estimated counterfactual production levels by installation,[18] we estimate the number of free allowances (EUA for EU Allowance, which is the official name of pollution permits traded in the EU Emissions Trading Scheme) received at the plant level under the various scenarios. As an example, let us consider a plant, which is functioning at 50% E/HEL and receiving 1 million EUAs.[19] Suppose that our econometric model finds that the counterfactual activity level of this plant is 40%. This plant would have received 0.4 million EUAs under OBA, 1 million EUAs under EX and EXALTG, 0.5 million EUAs under EXALTNG.

In this short example, we see that gaming from 40% to 50% allows obtaining 0.5 MEUAs more allowances, but involves 0.11 Mt CO_2 of additional emissions,[20] so that the net gain in terms of allowances is 0.39 MEUAs. To convert the various effects into monetary value, we assume a CO_2 price at 7.95€/t, which corresponds to the average future price (December 2013) during year 2012.[21] In our quantification of the net financial impact we consider that the increased production is sold at marginal cost, and so has no impact on profits. We refer to this hypothesis as H_3. In practice plants may actually sell their excess production at a different price, the

[18] As we perform a Monte Carlo analysis, there is not "a" counterfactual but 10,000. For simplicity, we will explain the reasoning as if there was just one (these different steps are simply repeated for each sample of counterfactual).

[19] Caution, in order to make computations easier, this plant does not have the same characteristics as the one in Section 2.2. They both have a clinker carbon intensty of 0.8 tCO_2 per ton of clinker, but the latter had a HAL at 1Mt per year, this one receives 1 million EUAs, which is equivalent to a HAL of 1.38 Mt per year.

[20] Assuming that the plant has a clinker carbon intensity of 800 kg CO_2 per ton of clinker.

[21] Source: ICE database (`http://data.theice.com/MyAccount/Login.aspx`).

Scenarios	Allocations	Production
OBA	Proportional to Activity $HAL \times ALCF \longleftrightarrow q$ in Eq (5.1))	Counterfactual (explained in section 5.2.3)
EX	Independent of Activity ($ALCF = 1$ in Eq (1))	Same as OBA
EXALTNG	Hybrid (Eq (5.1))	Same as OBA
EXALTG	Same as EXALTNG	Actual 2012 Production

Table 5.2: Scenarios

important point being that the associated revenue be higher than the associated inefficiency costs (see Section 2.2). The precise financial impact is bound to depend on circumstances specific to each plant which are unobservable. H_3 allows for an estimate of the financial impact.

In summary, for the four different scenarios, we compute production, emissions and allocation. The net allowances (allocations minus emissions) are compared for the scenarios EX, EXALTNG, EXALTG and OBA. Comparing other scenarios to OBA gives an estimation of overallocation profits (in MEAUs or M€). The difference between EXALTG and EXALTNG gives the impact of gaming. Table 5.2 summarises how allocations and production are obtained under each scenario.

Comparing counterfactual net exports to real net exports gives the part of the excess clinker production which is destined for clinker exports and cement exports. If no stockpiling is assumed, the remaining part can be attributed to the change in the clinker ratio.

5.2.3 Estimation strategy

We calculate counterfactual clinker production levels of a plant in 2012 and characterise output behaviour of firms conditional on national and plant level variables. As noted, the unobserved level activity of plant i in year t is approximated by the observed level of emissions $PlantActivity_{i,t} \approx \dfrac{E_{i,t}}{HEL_i}$ the activity level of plant i in year t (ratio of emissions divided by historic emissions level). As noted also, we assume that cement consumption is independent of allocation rules. Therefore, cement consumption would have been the same in 2012 had the ALTs rule not been implemented.

We use a multiplicative panel data model to estimate the following specification of clinker production level in plant *i* at time *t* to obtain parameters used to calculate counterfactual activity level in 2012:

$$\begin{aligned} \triangle lnPlantActivity_{i,t} = \alpha_0 + \beta_1 \triangle lnCementConsum_{c\ni i,t} \\ + \beta_2 \triangle lnGDP_{c\ni i,t} + \gamma_1 lnRelativeCO2Intensity_i \\ + \gamma_2 lnRelativePlantsize_i + \gamma_3 Coast_i + \varepsilon_{i,t} \end{aligned} \tag{5.3}$$

In order to accommodate the autoregressive nature of plant activity, we define all country-level variables (source of the data is in Table 5.3) including the dependent variable in first differenced terms. This allows us to difference out the time-invariant country specific heterogeneity, using adjacent observations. The dependent variable is the (first differenced) natural log of the activity level of plant *i* in year *t*. Cement consumption and GDP are also expressed in first differenced natural log terms. In addition, we include time invariant[22] plant-level variables: the relative average carbon intensity of a plant;[23] relative plant size;[24] and a dummy variables for coastal plants.[25] In order to minimize measurement errors which would bias the regression, we regroup some small countries into larger entities which are coherent in terms of regional market: Baltic countries, Benelux, Norway-Sweden and Slovenia-Italy. As the Breush-Pagan test reveals the presence of heteroskedasticity,robust standard errors clustered at the country level are used.

[22]Some variables such as the clinker carbon intensity of the plant may not be strictly time invariant but it is the case in the first approximation.

[23]The relative carbon intensity is defined as the natural log of carbon intensity at the plant level divided by the average carbon intensity in the country it is located ($RelativeCO2Intensity_i = ln(\frac{I_{HAL,i}}{\overline{I_{HAL,c\ni i}}})$, where $\overline{I_{HAL,c\ni i}}$ is the average carbon intensity of plants (in tons of CO_2 per ton of clinker) in the country where the plant *i* is located).

[24]This is defined as the natural log of the historical activity level of the plant divided by the average historical activity level in the country it is located ($RelativePlantsize_i = ln(\frac{HAL_i}{\overline{HAL_{c\ni i}}})$, where $\overline{HAL_{c\ni i}}$ is the average carbon intensity of plants (in tons of CO_2 per ton of clinker) in the country where the plant *i* is located).

[25]The dummy $Coast_i$ is equal to one if the plant is located near the coast (less than 50km, this was done thanks to the geolocalization of the plants in the EUTL data). It concerns 61 plants out of 246.

Variable	Source
Emissions and HEL	EUTL
Clinker net exports (NE_K)	Eurostat International Trade Data is originally given by country pairs. Total net exports are re-computed. Product category: "Cement Clinker" (252310)
Cement net exports (NE_C)	Eurostat International Trade Product category: Difference between "Cement, incl. cement clinkers" (2523) and "Cement Clinker" (252310).
Cement consumption (C_C)	1) Cembureau (2013) for the main European countries 2) VDZ (Table C10) for Baltic countries and Norway
Country GDP (GDP)	World Bank Development Indicators Database They are in billion current US dollars
Clinker production (Q_K)	1) EUTL-derived estimation (through estimated clinker carbon intensity and emissions, see section 5.5.1) 2) For checking, supplementary data were obtained from several sources *e.g.*: a) National cement association data when reliable and exploitable: - Spain (Oficemen, p90) - Germany (VDZ, table A2) - France (Infociments, Table p7) b) Getting the Numbers Right database, indicator 311a for available countries (UK, Italy, Poland, Czech Republic, Austria)

Table 5.3: Data Sources

	(1)
Log Cement Consumption	0.819***
	(0.113)
Log GDP	0.235
	(0.180)
Log Relative Carbon Intensity	-0.333***
	(0.0128)
Log Relative Historical Activity Level	0.0125
	(0.0114)
Coastal dummy	-0.037***
	(0.0128)
Constant	-0.003
	(0.00872)
Observations	737
Plant level fixed effects	No
R^2	0.21

Notes: * p<0.1; ** p<0.05; *** p<0.01. Robust standard errors in parenthesis clustered at the country level. The dependent variable is the first differenced natural log of plant activity level. The sample includes 246 clinker producing plants identified as operating between 2010 and 2012, across 26 EU Member States, for the years 2008-2011.

Table 5.4: Regression results of corrections at the plant level

Table 5.4 column (1) shows the results for the period 2008-2011 (post-crisis). Cement consumption has a statistically significantly effect on clinker production, with an estimated elasticity of 0.819 (hence if the demand at the country level decreases by 10%, the production at the plant level decreases by 8.19%). GDP is not statistically significant with an estimated elasticity of 0.235. The relative plant size is not significant. Conversely, the carbon intensity of the plant has a negative effect, suggesting that production is lower in the most carbon intensive plants. Finally, the parameter *Coastal* is statistically significant and also negative. Production in coastal plants is lower by 4% in average than in inland plants. We could also have expected the opposite (coastal plants producing more, *e.g.* their production declining less, in order to export). This could reflect a strategy of cement companies to diminish production in coastal plants in the long run.

As a robustness check, we also estimate a fixed effects model which include plant level fixed effects to control for time invariant unobserved heterogeneity of clinker production behavior. Parameter estimates from the fixed effects regressions are similar suggesting that the combination of country-level fixed effects (implemented by first differencing) and time invariant plant level variables do a good job at controlling for heterogeneity in our random effects estimation. A number of further robustness tests were conducted. For example, we additionally ran the same specification using the correlated random effects model (Wooldridge, 2010) and also tested the influence of other obtainable variables to predict clinker output including year dummies, lagged values, square terms. We found that the results were stable across the various estimators and specifications.

These parameters from column (1) are thus used to estimate counterfactual activity level. In order to give results robust to uncertainty, we use a semiparametric approach (Powell, 1994) by specifically modelling the multiplicative error. The counterfactual plant activity level is then not fixed but is a random variable:

$$
\begin{aligned}
\widetilde{PlantActivity}_i^{CF-2012} &= PlantActivity_i^{2011} \times exp(\hat{\alpha_0}) \\
&\times \left(\frac{CementConsum_{c\ni i,2012}}{CementConsum_{c\ni i,2011}}\right)^{\hat{\beta}_1} \times \left(\frac{GDP_{c\ni i,2012}}{GDP_{c\ni i,2011}}\right)^{\hat{\beta}_2} \\
&\times RelativeCO2Intensity_i^{\hat{\gamma}_1} \times RelativePlantsize_i^{\hat{\gamma}_2} \\
&\times exp(\hat{\gamma}_3 Coast_i) \times exp(\widetilde{\varepsilon})
\end{aligned}
\tag{5.4}
$$

Extending the smearing estimate of Duan (1983), we first fit the distribution of $\widetilde{\varepsilon}$ with a kernel density estimation like in Horowitz and Markatou (1996) which gives us its piecewise linear cumulative distribution function. The latter allows us simulating $\widetilde{\varepsilon}$ (which has a standard deviation of 14%) via inverse transform sampling. We perform a Monte Carlo simulation with 10,000 samples and report the average and the 95% confidence interval in Tables 5.5, 5.6 and 5.7.

5.3 Results

5.3.1 Impacts of ALTs on the plant distributions

Figure 5.2 displays the distribution of plant activity levels for 2012 (EXALTG), the counterfactual[26] production (EX, EXALTNG, OBA) and also the distribution in 2011 for comparison. In LD countries, there is a marked jump in installations operating around the 25% and 50% activity level thresholds in 2012, whereas the counterfactual distribution for these countries is not skewed at the thresholds. We find that in LD countries where 117 of the 246 cement installations are located, ALTs should have reduced free allocations in 50 of them, but due to gaming, only in 20 installations was it reduced in reality. Thus, in line with preliminary findings of Neuhoff et al. (2014), these results show clearly that cement companies have indeed altered plant production levels in response to ALTS rules. In MD countries, this response is noticeable but to a much less degree. The contrast between LD and MD shows the importance of the demand collapse in triggering this gaming

[26] There is not "a" but 10,000 versions of the counterfactual. The distribution displayed here corresponds to the central scenario (with average activity level for each plant).

	LD countries	MD countries	All countries	Spain	Greece
Production (CF)	47.2	80.2	127.4	12.4	3.6
in Mtons	[45.2,49.4]	[76.9,83.7]	[123.6,131.5]	[11.5,13.5]	[3.0,4.3]
Production (observed)	54.4	79.4	133.8	16.0	5.6
in Mtons					
Increased Production	+7.2	-0.8	+6.4	+3.5	+2.0
in Mtons	[5.0,9.2]	[-4.2,2.5]	[2.3,10.2]	[2.5,4.4]	[1.3,2.6]
	p=1.00	p=0.33	p=1.00	p=1.00	p=1.00
Increased emissions	+6.4	-0.6	+5.8	+3.1	+1.8
in Mtons CO_2	[4.5,8.2]	[-3.6,2.2]	[2.2,9.1]	[2.2,3.8]	[1.2,2.3]
	p=1.00	p=0.34	p=1.00	p=1.00	p=1.00

Note: Reported values are the average of the 10,000 simulations and the 95% interval. p is the probability that the value is above zero.

Table 5.5: Production and Emissions for the observed (EXALTG) and counterfactual (EX, OBA, EXALTNG) scenarios

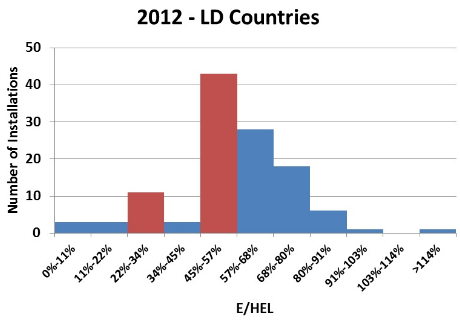
behaviour.

5.3.2 ALTs impacts on clinker production and emissions

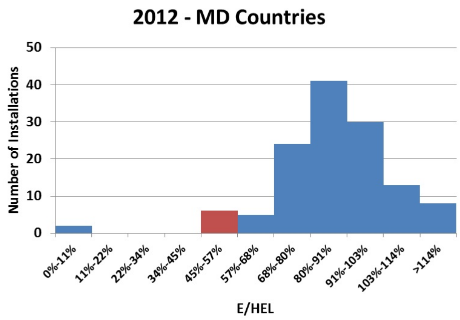
Table 5.5 gives the clinker production and the emissions for 2012 (EXALTG) and the counterfactual (EX, EXALTNG, OBA). The excess clinker production due to the introduction of thresholds rule is quantified. It represents an increase of 15% (+7.2Mt) in LD countries, 28% (+3.5Mt) for Spain and 56% (+2.0Mt) for Greece. These increases are extremely large, even if the global impact at the EU level is more modest (5%). The increase in the clinker production translates into increases in emissions. Altogether we estimate that an additional 5.8 Mt CO_2 (+5% for the sector as a whole) have been emitted by EU cement firms as a consequence of the strategic behaviour of cement companies.

5.3.3 Impact of gaming on plant distribution on the free allowances

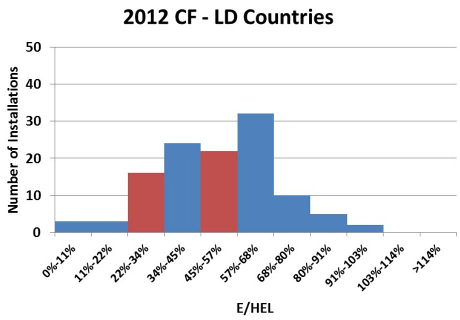
Table 5.6 gives the amount of EUA's that are allocated to cement installations under the four scenarios (EX, EXALTNG, EXALTG, OBA). If installations received



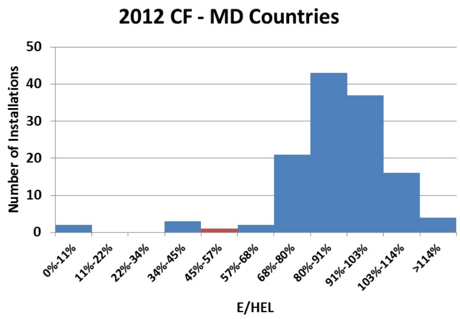
(a) 2012 LD



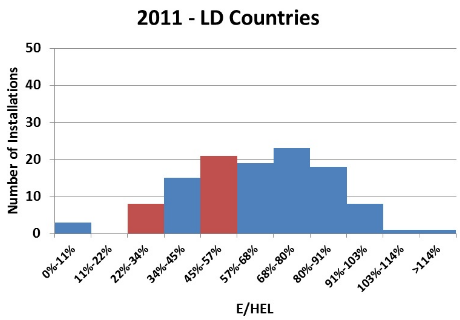
(b) 2012 MD



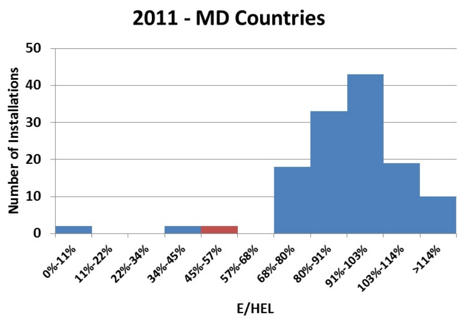
(c) 2012 Counterfactual LD



(d) 2012 Counterfactual MD



(e) 2011 LD



(f) 2011 MD

Note: An appropriate use of 2013 allocation data enables us to indirectly distinguish installations that have been in 2012 above or below thresholds (25% and 50% of q/HAL). We find that whenever E/HEL is superior to 45% (respectively 22%), the corresponding installation is above the first (respectively second) activity level threshold (see appendix 5.5.1 for more explanations).

Figure 5.2: Distribution of installations according to their activity level (approximated by E/HEL) in 2012 for observed and counterfactual production. 2012CF stands for counterfactual of 2012. Red bars indicate categories just above thresholds

	LD countries	MD countries	All countries	Spain	Greece
EXALTNG	55.1	68.1	123.2	14.9	4.3
	[52.8,57.3]	[67.2,68.9]	[120.8,125.6]	[13.5,16.3]	[3.5,5.1]
EXALTG (observed)	68.4	69.6	138.1	20.7	7.3
OBA	36.1	62.2	98.2	9.5	2.7
	[34.5,37.7]	[59.6,64.9]	[95.2,101.5]	[8.7,10.2]	[2.2,3.2]
Allowances Gaming Gain	+13.3	+1.5	+14.8	+5.8	+3.0
	[11.1,15.6]	[0.7,2.4]	[12.5,17.3]	[4.4,7.2]	[2.2,3.8]
	p=1.00	p=1.00	p=1.00	p=1.00	p=1.00
Net Gaming Gain (minus Emissions)	+6.9	+2.1	+9.0	+2.8	+1.2
	[4.9,9.0]	[-0.5,5.0]	[5.7,12.5]	[1.7,3.8]	[0.6,1.8]
	p=1.00	p=0.94	p=1.00	p=1.00	p=1.00

Note: Reported values are the average of the 10,000 simulations and the 95% interval. p is the probability that the value is above zero.

Table 5.6: The Free Allowances (MEUAs) under the four scenarios

100% of their allowances regardless of their activity (*i.e.* the allocation under the EX scenario), then LD countries and MD countries would have received 74.5 and 70 million EUAs respectively. OBA allocations would lower allocations to 36.1 and 62.2 million EUAs respectively. The decrease in allocations is more significant for LD countries because the average activity is much lower.

As explained, the scenario EXALTNG can be seen as an imperfect approximation of the OBA rule. If there had been no gaming, it would have set the allocations at 55.1 and 68.1 million EUAs. Thus for the cement sector as a whole, ALTs reduced overallocation in 2012 by 6.4 MEUSs compared to the scenario without ALTs. Had OBA been implemented instead, overallocation would have been further reduced considerably by 40 MEUAs, which corresponds to 29% of the total cement sector free allocation in 2012. The effect for the MD countries is negligible, as most of installations have an activity level superior to 50%. However for LD countries the theoretical effect of the threshold rule as an approximation of the OBA rule would have been more significant: a 50% (that is (74.5 – 55.1)/(74.5 – 36.1)) reduction should have been obtained. With gaming (EXALTG) a reduction of only 16% prevails (that is (74.5 – 68.4)/(74.5 – 36.1)). For Spain the percentages would respectively be 61% and 20%; and for Greece 73% and 24%. Further, we estimate the allowances gaming gain at 14.8 MEUAs, located almost exclusively in LD countries, and a net gaming gain (deducing extra emissions) of 9.0 MEUAs.

5.3.4 Financial potential gain associated with gaming

In the calculation of the potential gain we assume that the increased production is sold at marginal cost, and so has no impact on profits. This gives an upper bound for the profits that could be achieved with gaming since it does not take into account the possible inefficiency costs: logistics cost for production shifting, extra sales expenditures and rebates for increased exports, opportunity cost for increasing the clinker to cement ratio). That there are inefficiency costs can be seen from the fact that not all plants achieved the 50% threshold, but some gaming was certainly worthwhile since a large proportion of plants did manage to get to the target. To convert the increase in free allowances and the increase in emission rights into monetary value, we need to assume a CO_2 price. It should be clear that the amount of profitable gaming is dependent on the CO_2 price. We shall come back to this point in our discussion of the results. Table 5.7 gives the potential profit associated with gaming for a CO_2 price at 7.95€/t, which corresponds to the average future price (December 2013) during year 2012. Then it reflects more expected gains than actual gains, which may be lower or higher (the CO_2 price decreased the following year, but firms may have banked these extra allowances and the CO_2 price may rise in the future).

For LD countries, the potential gain of EX relative to OBA is estimated through the net increase of allowances which is 74.5 – 36.0 Mt CO_2 and a EUA price 7.95€/t which makes 306 M€. With the introduction of the threshold rule this increase would have been only 158 M€had the firms not gamed the scheme. The reduction is coming from the reduced amount of free allocations due to the downfall in market demand. The gaming increases the amount of free allocations but increases emissions, bringing a potential gain at 213 M€, which represents an increase of 35% (+55M€) relative to 158 M€. For Spain the per cent increase is

Millions of € relative to OBA	LD countries	MD countries	All countries	Spain	Greece
EX	306	62	368	113	48
	[292,318]	[40,83]	[342,392]	[107,119]	[44,52]
EXALTNG	158	49	207	50	13
	[145,170]	[27,69]	[181,231]	[44,55]	[9,16]
EXALTG	213	66	278	72	23
	[209,216]	[65,67]	[276,281]	[69,74]	[22,24]

Note: Reported values are the average of the 10,000 simulations and the 95% interval.

Table 5.7: Quantification of the monetary value of excess free allocations for the various scenarios.

44% (+22M€) and for Greece it is 77% (+10M€). These figures are substantial even though the carbon price was low at that time. This explains why firms undertake the various inefficiencies described earlier to capture part of this gain.

5.3.5 Where does the excess clinker end up? Indirect evidence revisited

This section revisits the indirect evidence of excess clinker production proposed by (Neuhoff et al., 2014). As noted, three channels have been identified, production shifting, exports increase and clinker ratio increase.

Production shifting in multi-plants companies

Cement company executives in interviews reported that subsequent to the introduction of ALTs, it was frequent practice to arrange production levels across plants to ensure being above the threshold at as many units as possible (Neuhoff et al., 2014). We observe output behaviour consistent with these statements in several cement companies which have a number of plants producing close to the thresholds. Table 5.8 presents four examples. [27] In each of these firms in 2012, production (within the same geographical country) simultaneously falls in one plant

[27] We only display here groups of installations belonging to a country-company that are the most consistent with production shifting, but avoid cherry-picking individual installations. For the four cases, all installations of a certain country-company are displayed.

Country-Company	Installation	E/HEL 2011	E/HEL 2012
Greece-W	1	34%	49%
Greece-W	2	77%	66%
Greece-W	3	11%	0%
Spain-X	1	42%	50%
Spain-X	2	57%	46%
Spain-X	3	68%	56%
Hungary-Y	1	41%	46%
Hungary-Y	2	68%	50%
Portugal-Z	1	34%	64%
Portugal-Z	2	55%	51%
Portugal-Z	3	71%	60%

Note: An appropriate use of 2013 allocation data enables us to indirectly distinguish installations that have been in 2012 above or below thresholds (25% and 50% of q/HAL). We find that whenever E/HEL is superior to 45% (respectively 22%), the corresponding installation is above the first (respectively second) activity level threshold (see appendix 5.5.1 for more explanations).

Table 5.8: Evidence of within-firm-country production shifting to meet thresholds.

(which produced well above the threshold in 2011), and rises in another plant above the threshold (which was previously operating below the threshold).

Exports

Table 5.9 gives net exports of clinker and clinker embedded in cement from 2010 to 2012 for LD and MD countries. We observe a surge in clinker net exports in LD countries: 6.21 Mt in 2012, compared to 2.03 Mt and 1.94 Mt in 2010 and 2011 respectively. In contrast MD countries remained small net importers of clinker and no significant shift was observed in their trade patterns. Further analysis revealed that these clinker exports in 2012 were destined mainly to countries in Latin America and Africa, including Brazil, Togo, Ghana, Cameroon, Côte d'Ivoire, and Mauritania and Nigeria.

LD Countries	2010	2011	2012
Clinker	2.03	1.94	6.21
Clinker in Cement	5.49	4.58	6.37
MD Countries	**2010**	**2011**	**2012**
Clinker	-0.93	-0.74	-0.71
Clinker in Cement	2.24	2.46	2.02

Note: Source: Eurostat. We use a common clinker ratio of 75% to compute clinker embedded in cement.

Table 5.9: Clinker net exports in 2010, 2011 and 2012 in LD and MD countries in millions of tonnes.

Clinker Ratio	2010	2011	2012
MD Countries	76%	76%	77%
LD Countries	74%	72%	74%
Spain	79%	76%	82%
Greece	76%	71%	75%

Table 5.10: Clinker-to-Cement Ratio in selected areas (source: authors' analysis).

Clinker ratio

Another way excess clinker production might materialise is in a higher clinker-to-cement ratio. That is, firms could use more clinker to produce the same amount of cement. The clinker ratio can be recomputed at the macro level (state of group of states) with the formula $R = \dfrac{Q_K - NE_K}{C_C + NE_C}$, where Q_K is the clinker production, NE_K and NE_C net exports of clinker and cement, and C_C the cement consumption (see Appendix 5.5.2 for explanation and Table 5.3 for data source). Table 5.10 shows the clinker ratio for the MD countries, LD countries, Spain and Greece. The historical declining trend in the clinker-to-cement ratio has reversed in 2012.

Region	Total Increase in Clinker Production	2012 Clinker Net Exports			2012 Cement Net Exports			Clinker Ratio	
		CF	Observed	Diff	CF	Observed	Diff*R	Effect	Relative
All LD	7.2	4.6	6.2	+1.6	6.1	8.5	+1.7	3.9	6%
All MD	-0.8	0.4	-0.7	-1.1	3.3	2.7	-0.4	0.7	1%
All	6.4	5.0	5.5	+0.5	9.4	11.2	+1.3	4.6	3%
Spain	3.5	2.2	3.4	+1.2	2.2	2.6	+0.3	2.0	12%
Greece	2.0	0.5	1.8	+1.3	1.5	1.7	+0.2	0.5	9%

Table 5.11: Routes of excess clinker production decomposition.

5.3.6 Decomposing the channels for clinker disposal

In order to better understand the effects of the distortions that arise from ALTs, we attempt to decompose the excess clinker output[28] into the main destinations to which they are channelled through: changes to clinker ratio of domestic cement and increase in exports (clinker or cement). Although it is likely that there is some stockpiling, the lack of data makes it difficult to attribute excess production to this channel.

This decomposition requires that actual net export volumes of cement and clinker are compared to counterfactuals levels (see Appendix 5.5.3 for the estimation method and data used). Assuming no stockpiling, we can attribute the remaining excess clinker output to clinker ratio increase. Table 5.11 gives the results. Figure 5.3 provides a graphical representation. For LD countries, net exports of clinker increased by 6.2 Mt while our counterfactual is 4.6 Mt (+1.6 Mt); the net export of cement increased by 8.5 Mt while the counterfactual is 6.1 Mt (+1.7Mt of clinker embedded); this implies that 2.4 Mt of clinker went into the increased content of clinker in cement. This latter figure represents an increase of 6% relative to our counterfactual for the clinker to cement ratio as defined in the previous section. The values of clinker ratio effect are higher here than the estimates in Section 4.6 suggesting that stockpiling of excess clinker output may be occurring, as well as increased clinker ratio of cement exports.

[28]Production shifting in multiplant companies, which does not generate excess clinker output, is not quantitatively assessed.

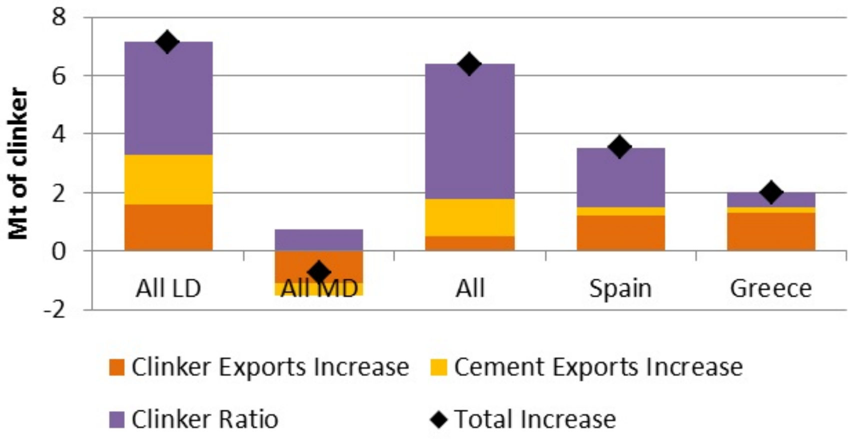


Figure 5.3: Routes of excess clinker production decomposition.

5.3.7 Discussing the impact of hypotheses H1 and H2 on results

In this section we will discuss how results (mainly Tables 4 to 6) are modified if H_1 or H_2 does not hold.

Scenario EXALTG corresponds to real observations, so emissions and allocations are never modified. However, changes in hypotheses potentially modify the counterfactual scenarios EX, EXALTNG (which by construction corresponds to EX with only a different allocation rule) and OBA.

If H_1 does not hold ($\overline{H_1}$), it implies there is a carbon price pass-through in EX/EXALTNG, but not in OBA. Then, in $\overline{H_1}$, clinker production (and thus emissions) would be higher in OBA compared to EX/EXALTNG, because consumption is higher due to a price effect and also because of a better protection against carbon leakage. Several papers showed that OBA acts as a production subsidy (Fischer, 2001; Fischer and Fox, 2007). Let us call $+\delta E_1$ the corresponding emissions increase, which depends on many factors (including the price of carbon, price elasticity and regional competition).

Second, if H_2 does not hold ($\overline{H_2}$), a part of the excess clinker production due to gaming would flood the local market. This positive shock in supply would lower prices and increase consumption. Then the observed consumption (EXALTG scenario) would be higher than if the threshold rule had not been implemented (EX, EXALTNG and OBA). Since we base our estimation of production

	Table 5.5 **ΔEmissions** **EXALTG *vs.*** **EXALTNG** ***(Extra Pollution due to Gaming)***	**Table 5.6** **ΔProfits**[1] **EXALTG *vs.*** **OBA** ***(Overallocation Profit)***	**Table 5.7** **ΔProfits**[1] **EXALTG *vs.*** **EXALTNG** ***(Extra Profits due to Gaming)***
$\overline{H_1}$: Pass-through in OBA *vs.* EX/EXALTNG	0	$p_{EUA} \times$ $[\delta E_1 - \delta A_1^{OBA}]$ $(\gtrsim 0)$	0
$\overline{H_2}$: Surplus of production due to gaming poured into local market	$+\delta E_2$	$p_{EUA} \times$ $[\delta A_2^{OBA} - \delta E_2]$ $(\lesssim 0)$	$p_{EUA} \times$ $[\delta A_2^{T} - \delta E_2]$

[1] Only related with allowances surplus, not with margins
Explanations: in $\overline{H_1}$ for the OBA scenario, emissions are increased by $+\delta E_1$ compared to our estimates, involving an increase in allocation $+\delta A_1^{OBA}$ (proportional to the increased production and the clinker benchmark). In $\overline{H_2}$, for the EX, EXALTNG and OBA scenarios, emissions are decreased by $-\delta E_2$ compared to our estimates, involving a decrease in allocation $-\delta A_2^{OBA}$ in OBA and $+\delta A_2^{T}$ in EXALTNG.

Table 5.12: Change in main results with $\overline{H_1}$ and $\overline{H_2}$

on consumption, using this reduced consumption level instead of the observed one would involve that in $\overline{H_2}$, the estimated clinker production level in EX, EXALTNG and OBA would be lower than in our results. Let us call $-\delta E_2$ the corresponding decrease in emissions, which is highly differentiated among regions (more important in low demand countries).

Table 5.12 sums up how three main results are modified with $\overline{H_1}$ and $\overline{H_2}$: extra pollution due to gaming, overallocation profits and extra profits due to gaming.

First, extra pollution due to gaming is not modified by $\overline{H_1}$ but is increased by $\overline{H_2}$. That is, we would have underestimated extra production due to gaming in our main results, because our estimated counterfactual production would have been overestimated, being based on too high a consumption level (the "true" one would have been lower because no cement would have been flooded the local market driving down the prices).

Second, overallocation profits are modified but only at the margin (they slightly increase with $\overline{H_2}$, and slightly decrease with $\overline{H_2}$). [29] Thus, when both assumptions are relaxed, the total effect is even smaller because directional effects go in opposite directions.

Third, extra profits due to gaming are not modified in $\overline{H_1}$ but potentially in $\overline{H_2}$. Results would never be modified significantly, the nature of the change being ambiguous but more likely to be upward in countries where gaming occurred.[30] Our figures would then bring a low bound estimation of the profits increase due to gaming. The main idea is that because clinker production in EX is lower in $\overline{H_2}$ than in H_2, we may have underestimated among plants above the threshold in EXALTG the number of plants that were below the threshold in EX.

In summary, our results are robust in relaxing H_1 or H_2. The most controversial assumption, H_1, in particular, has limited impact on the results. Furthermore, qualitative assessment suggests that if anything, our results would be biased in the conservative direction (underestimation of extra production and extra profits due to gaming).

5.4 Conclusions and policy options

An important change in the EU-ETS phase 3 for EITE concerns the introduction of the activity level threshold rule (ALTs). The underlying rationale for its introduction is that it would reduce the overallocation profits in case of downfall in the

[29] Indeed, $\delta E - \delta A^{OBA} \approx (I_A - I_B)\delta Q$ (where $(I_A - I_B)$ is the difference between the average and the benchmark clinker carbon intensity), which is at least an order of magnitude lower than the original overallocation profits. As an example, let us consider a country producing 18 Mt of clinker (30 Mt in HAL), so if $\delta Q = 0.1$ or 2, $\delta E - \delta A^{OBA} = 0.01$ or 0.2 EUA with $(I_A - I_B) \approx 0.1$. In comparison, original overallocation profits are at least 5 MEUAs.

[30] $\delta A_2^T - \delta E_2 = \delta A_2^T - \delta A_2^{OBA} + \delta A_2^{OBA} - \delta E_2$. $\delta A_2^{OBA} - \delta E_2$ is negative and small (cf previous remark). Further, at the plant level, $\delta A_2^T - \delta A_2^{OBA}$ is negative and small if the shock in production does not cross an activity level threshold and positive and big when it does (which is more likely to happen in low demand countries). At the aggregated level, the change is then likely to be positive in countries where gaming occurred (lower reference of production and bigger production shock). However, if we suppose that the majority of the production surplus is not poured into the local market, the change is still significantly lower than the original value (which is roughly $\Delta A^T - \Delta E$, *e.g* the same but with a bigger shock in production).

demand: whenever the activity level of an installation falls below some threshold (50%, 25%, 10%) relative to its historic activity level used to allocate free allocations, the allocation would be reduced accordingly (50%, 25%, 0%).

Our ex post analysis of year 2012, the first year in which the threshold rule applies, focused on the cement sector, a sector in which approximately half the EU countries had experienced a significant downfall in consumption (LD countries). It provides a natural experiment to evaluate the consequences of this rule.

Our main conclusion is that while ALTs did reduce to some extent overallocation profits, it also created operational distortions which lead to outcomes inconsistent with the low carbon transition of EU energy intensive industries. The reduction in overallocation profits is less than expected because of the gaming behaviour of the industry to achieve the thresholds, during periods of low market demand. Thanks to the elaboration of a counterfactual, we have been able to quantify that after the introduction of ALTs: the potential overallocation profit with gaming is 278 M€(2 €/t clinker) and 207 M€without gaming, while it would have been 368 M€in the absence of ALTs. The expected reduction in windfall profits due to the ALTs is 44% while the actual reduction is 24%. The incentives are magnified in low demand countries, where profit with gaming is 213 M€(3.9 €/t clinker) and 158 M€without gaming, while it would have been 306 M€without ALTs. We examined three ways in which firms' operations are altered in response to ALTs: shifting production among plants, increasing net exports of clinker and cement, increasing the clinker to cement ratio.

In the 2000's top management attention on the issues of climate change emerged as an important dimension of corporate social responsibility and a large number of companies got involved in proactive strategies to limit their own emissions (Arjaliès et al., 2011). The EU-ETS positively contributed to turn this strategy into operational practise by putting a price on carbon. To put it simply, we observed a progressive alignment all through the firm between corporate social responsibility (CSR) and the carbon mitigation objectives of the EU-ETS. The operational distortions reported in our study due to the introduction of ALTs are particularly detrimental in this respect: the production shifting goes against the restructuring of the assets to achieve scale economies, a key factor of cost efficiency in cement; the increased exports induce some relocation of foreign cement production in the

EU, generating cost inefficiencies and extra emissions due to transportation; the increase in the clinker to cement ratio goes against one of the main drivers to limit emissions in cement production (see Chapter 4). In short, distortions generated by the introduction of the ALT have hindered the progressive alignment of incentives away from the low carbon transformation in this sector.

Our results have been obtained in a context of low carbon price, severe downfall in market demand, and large free allowance allocations. However, a higher carbon price would make our results even more relevant; the higher the carbon price the higher the incentive to achieve the thresholds.[31] Had we observed growth, the threshold rule may have been less relevant. Anecdotal evidence[32] suggests that instead, the reserve for new entrants may have been a more important source of distortions .(there would have been an incentive to have a artificially high production during the period used to fix the equivalent of HAL for new entrants)

These considerations suggest that the activity level thresholds may need to be reconsidered for sectors such as cement for which carbon costs represent a significant share of production costs. This raises the question of what to put in its place instead. As mentioned in the introduction economists generally agree that, in the absence of global carbon prices, replacing free allocation with full auctioning and using border carbon adjustments offers the most efficient solution. This is because it helps in levelling the carbon costs between domestic and foreign producers while also allowing for carbon costs to be passed along the value chain to incentivise demand side abatement. Politically this solution has not yet gained serious traction. This is largely due to concerns that border-levelling may be perceived as protectionism disguised as environmentalism and hence not conducive to building trust in international climate negotiations. However, the situation may change. If one looks forward to the post-2020 period, a larger number of nations are expected to have begun implementing carbon prices. More countries will face similar challenges related to designing appropriate anti-leakage measures that

[31]Take a EUA price at 20€/t a simple extrapolation for LD countries would bring up the potential wind fall profit to 236*20/9 = 524 M€. However if we assume that all plants achieve the 50% threshold, a reasonable assumption for a EUA price at 20€/t, it would go up to 583 M€. The expected reduction remains at 42% but the actual one drops to 22%. Note however that a high carbon cost might endanger the validity of assumption H_1 and could possibly lead to a result in which EXALTG would be preferred to EX, but still worse than OBA.

[32]Ref. private conversation with industry representatives.

the EU now faces and thus there may be more scope for cooperative approaches. Border-levelling via international cooperation would, however, take time to negotiate and design. This raises the question as to the interim solution.

One option is to increase the number of activity level thresholds to reduce the incentive to game output. For example, a threshold at 50%, 60% and 70% for cement may incentivise a larger number of installations to increase their clinker production to the next highest threshold. Since thresholds create an allocation system that falls between an *ex-ante* and *ex-post* scheme, it would be much simpler to implement full output-based allocation for sectors like cement where the risk of distortions arising is high, because carbon costs are high relative to production costs in the absence of free allocation. The analysis in this chapter suggests that this option would outperform both ex ante allocation with and without thresholds in terms of reducing distortions and overallocation profits.

However, a number of issues must be carefully considered before going in that direction. A central drawback of a move to OBA is that little can be expected in terms of carbon price pass-through to product prices and hence demand side substitution towards lower-carbon goods. For sectors where carbon costs are high as a share of production costs, such as cement, this would significantly limit the EU's potential to reduce emissions cost-effectively and to decarbonise these sectors. Unlike ex ante allocation, OBA implies the loss of an absolute cap for free allocations and this may be politically contentious point. Further, the implementation of OBA to select sectors but not all may also raise political difficulties. There are ongoing discussions on how to circumvent these issues. For example the loss of demand side substitution incentives could perhaps be restored with a consumption charge on downstream products (Neuhoff et al., 2014). Output based scheme with hybrid benchmark has been implemented in California in 2012. An ex post study on this implementation would be welcome to see if, again, the devil lies in the details.

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5.5 Appendix

5.5.1 EUTL Data computations

Determination of the Activity Level Correction Factor of 2013 at the plant level

The key challenge is to correctly distinguish installations that are above or below thresholds (25% and 50% of q/HAL), despite the limitation that activity levels have to be approximated using emissions data (E/HEL). To do so, we exploit the observations from the 2013 allocation data, which revealed whether or not the installation had seen its allocation reduced because its 2012 activity level fell below a threshold. Allocations in 2013 are equal to (cf equation (5.1)):

$$A_{i,2013} = CSCF_{2013} \times I_B \times HAL_i \times ALCF_{i,2013} \tag{5.5}$$

Where $CSCF_{2013}$ is the 2013 Cross Sectoral Correction Factor (0.9427), I_B the clinker carbon intensity benchmark (766 kg CO_2 per ton of clinker), and HAL_i the Historical Activity Level of installation i (in tons of clinker). Transforming the previous equation, where both HAL_i and $ALCF_{i,2013}$ are unknown, we obtain:

$$\frac{CSCF_{2013} \times \frac{I_B}{I_A} \times HEL_i}{A_{i,2013}} = \frac{1}{ALCF_{i,2013}} \times \frac{I_{i,HAL}}{I_A} \tag{5.6}$$

Noting $I_{i,HAL} = \frac{HEL_i}{HAL_i}$ (corresponding approximately to the clinker carbon intensity for the HAL producing years), and I_A is the average clinker carbon intensity (863 kg CO_2 per ton of clinker, GNR, indicator 321) in 2008.

The ratio at the left part of the equation can be computed with available data. On the right part, we have $ALCF_{i,2013}$, which we want to find, and the ratio, $\frac{I_{i,HAL}}{I_A}$, which is unknown as well but bounded and likely to be close to 1. Indeed, $I_{i,HAL}$ varies in an extreme range from 720 kg CO_2 per ton of clinker to 1300 kg CO_2 per ton of clinker (and for the very large majority of the plants from 780 to 950 kg CO_2 per ton of clinker), which translates into a ratio $\frac{I_{i,HAL}}{I_A}$ varying from 0.83 to 1.51 (and most likely from 0.90 to 1.10). Then, if the ratio, is comprised between

0.83 to 1.51 (respectively between 1.67 and 3.01, and between 2.64 and 4.80[33]), we infer that $ALCF_{i,2013} = 1$, (respectively 0.5 and 0.25).

This enabled catching out situations in which imperfections in the E/HEL measure as a proxy for the q/HAL would have led to a false conclusion about whether an installation was truly above or below its activity threshold in 2012. We found that the actual thresholds for the E/HEL measure that matched the 2013 allocation data were slightly lower in practice, at 22% and at 45%, rather than 25% and 50%. Discussion with industry experts revealed that there was a logical explanation for this systematic bias: clinker producers often have more than one kiln inside an installation that is treated as a single unit for free allocation purposes. When demand falls, it is common to concentrate production in the most efficient kiln(s). Thus emissions may fall by slightly more than overall clinker production, creating a slight downward bias in E/HEL as a measure of q/HAL in low demand countries. This bias could also be explained by the clinker carbon intensity improvement between HAL years and 2012.

Determination of clinker carbon intensity and production at the plant level

Once the $ALCF_{i,2013}$ has been determined at the plant level i (see previous section), the plant clinker carbon intensity for HAL years, $I_{i,HAL}$, can then be obtained with the previous equation. For 20 plants (out of 246), we found an unusual number (below 700 kg CO_2 per ton of clinker), possibly due to a capacity increase, and put instead a default value equal to I_A. We also set the default value I_A when $A_{i,2013} = 0$ (meaning $ALCF_{i,2013} = 0$ or plant closure), making the computation impossible (15 plants). We then correct the first approximation of clinker carbon intensity so as weighted average[34] clinker carbon intensity in big countries corresponds to GNR data in 2008 (818, 831, 832, 797, 847, 858, 849 and 842 kg CO_2 per ton of clinker for respectively Austria, Czech Republic, France, Germany, Italy, Poland, Spain and the United Kingdom). Finally we correct values of clinker carbon intensity in plants of other countries in the same way, so as the European

[33] In our data there is actually a gap between 2.14 and 4.01 so no case of overlapping.

[34] The weights are production, as multiplying plant emissions by this first approximation of clinker carbon intensity gives a first approximation of clinker production at the plant level ($\widetilde{Q}_{K,i,2008} = I_{i,HAL} \times E_{i,2008}$).

weighted average clinker carbon intensity (I_A). Once clinker carbon intensity is estimated for each plant, clinker production can be obtained through emissions ($Q_{K,i,t} = E_{i,t} \times I_{i,HAL}$). We assume that clinker carbon intensity does not evolve over time.

5.5.2 Macro data consistency at the national level

If we denote the six different variables:

- Q_K clinker production
- Q_C total cement production
- NE_K clinker net exports
- NE_C cement net exports
- C_C cement consumption
- R clinker-to-cement ratio

We have two equations translating the conservation of cement on the one hand and the conservation of clinker on the other hand (neglecting stockpiling):

$$Q_C = C_C + NE_C \tag{5.7}$$

$$Q_K = R \times Q_C + NE_K \tag{5.8}$$

These equations must be verified for each country every year (for real of counterfactual scenario). In this chapter for real data, Q_K, NE_K, NE_C and C_C are obtained through different sources (see Table 5.3), and Q_C and R are re-computed (we have $R = \dfrac{Q_K - NE_K}{C_C + NE_C}$).

5.5.3 Counterfactual country level net exports of clinker and cement estimation

Counterfactual net exports of clinker and cement for each country are necessary to assess the channels of clinker disposal. A comprehensive analysis was not possible given the available data, and instead we use a simple first differenced estimation to control for country-level fixed effects and include cement consumption as the main explanatory variable. [35] This enables us to essentially extrapolate historic next export trends, whilst accounting for the influence of annual variation in cement consumption. The parameters are obtained from the following regression using data for the years 2008-2011 and 20 countries:

$$\Delta NE_{K,c,t} = \lambda_0 + \lambda_1 \Delta CementConsum_{c,t} + \varepsilon_{c,t} \tag{5.9}$$

$$\Delta NE_{C,c,t} = \mu_0 + \mu_1 \Delta CementConsum_{c,t} + \varepsilon_{c,t} \tag{5.10}$$

For clinker net exports, the coefficient on λ_1 is -0.162 and this is significant at the 5% level. Hence on average, if cement consumption decreases by 1 Mt, clinker net exports increase by 0.16 Mt. The negative sign on λ_1 is inline with expectations. The fit is good for the clinker net exports (R^2=0.41). For net cement exports, the coefficient on the cement consumption term is 0.025 and is not statistically significant at conventional levels. Changes in cement consumption thus do not predict changes in cement net exports and in this case the counterfactual is an extension of historic trends only. For a region c, we then compute counterfactual net exports as follows:

$$NE_{K,c,CF-2012} = NE_{K,c,2011} + \hat{\lambda}_1 \Delta CementConsum_{c,2012} \tag{5.11}$$

and counterfactual next exports of cement as:

$$NE_{C,c,CF-2012} = NE_{C,c,2011} + \hat{\mu}_1 \Delta CementConsum_{c,2012} \tag{5.12}$$

[35] As suggested by the Hausman test (if p-value are low, fixed effects are preferred), we used a fixed effect model. As the modified Wald test reveals the presence of heteroskedasticity, we present robust standard errors.

Conclusion

The first part of the thesis helped to clarify the significant research undertaken on carbon leakage and competitiveness, especially the modelling literature numerically assessing carbon leakage and the effectiveness of BCAs. The main message is that though positive, carbon leakage remains low and would not justify postponing action for the sake of environmental efficiency. Further, BCAs would be efficient to level the carbon playing field, but some leakage would remain because of the channel of international fossil fuel prices.

The second part of the thesis makes an important finding that the EU ETS has not induced operational carbon leakage in its first seven years of functioning. The interpretation of such a result remains problematic in terms of policy making. If one believes that companies consider the full opportunity costs of allowances, it means that carbon leakage is irrelevant for the observed range of carbon price; but it also means that free allocation was inappropriate to protect these sectors from carbon leakage in the first place (it would have only be useful to increase their profitability). However industry executives in their interviews (Neuhoff et al., 2014b) go against the economic theory that would predict an internalization of the carbon costs in the decisions of companies. The main reason would be strategic: to avoid killing the goose that lays golden eggs, the industry would have avoided actions contradicting the narrative that free allocation was necessary for leakage protection.[36]

This thesis cannot conclude on the effectiveness of free allocation to prevent carbon leakage. However, the last part of the thesis demonstrated that potential

[36] Such a narrative is credible only for an oligopolistic sector (to counteract free-riding incentives) and under low carbon price.

adverse effects on competitiveness were overcompensated at the expense of the credibility of the whole system. We showed that the EU ETS has not triggered significant abatement in the cement sector, but this could be true for most industrial sectors. Other studies suggest that it would have not induced significant innovation and low-carbon investments either (Laing et al., 2014). Generous free allocation and the authorization of offset credits for cost-containment purposes have helped build up a massive allowance surplus. This surplus, combined with the uncertainty of the future of the ETS, has depressed the carbon price to a worrying point. The economic downturn, which has been the main source of the surplus of allowances, could not have been predicted in the first place. However, concerns about adverse effects on competitiveness have gone against any attempts to fix the EU ETS after it entered into a crisis. The EITE industries obtained that the reference production giving the amount of free allocation for phase III would be the pre-crisis production level instead of a more realistic one.[37] Furthermore, out of the six options to fix the EU ETS presented in 2012 at the beginning of the crisis (European Commission, 2012), the less ambitious one was finally adopted (restricted use for international credits), while other options intended to raise the ambition of the EU ETS before 2020, such as an auction price floor or measures permanently removing allowances from the market, were discarded.

After criticizing the current design of carbon leakage provisions in the EU ETS, we present in the rest of the conclusion possible reforms. An adequate policy package for EITE sectors would promote low carbon transition without generating carbon leakage, avoid distortions among plants or sectors, and remain at low administrative costs. We believe that full auctioning should remain the reference for reasons of efficiency, fairness, transparency and simplicity (Zetterberg et al., 2012). Auctioning avoids the elaboration of complex rules in which companies exploit the asymmetry of information with the regulator to obtain the highest rent possible. All installations are treated equally and public revenues are maximized. Full auctioning for the electricity sector, which is not exposed to carbon leakage, makes common sense. It is unfortunate that it took seven years for the European Com-

[37] In Cembureau (2011) 2010 activity report we can read in p.2: "Cembureau and other Energy Intensive Industries were helped by DG Enterprise in securing that the recession years – 2009 and 2010 – will not be taken into account [...] to determine, *ex ante*, the number of allowances to be allocated for free. [...] This was another achievement.".

mission to enforce it, allowing in the meantime the power sector to reap colossal windfall profits by passing through the cost of allowances (Sijm et al., 2006; Fabra and Reguant, 2014). For the EITE sectors however, the unevenness of climate policies in the post-2020 regime would call for some type of adjustments in case of full auctioning. We do not consider that the difficulties of implementation are a good reason to discard BCAs prima facie. However, because a failure of implementation would undermine the system, a robust policy should be set for a transitory period. To prepare for the implementation of BCAs, an information-based policy displaying the carbon footprint of products to consumers could be launched to build expertise in carbon content measurement methods and raise awareness among consumers.

Because of the massive distributional issues generated by pure *ex ante* allocation, we favour output-based allocation despite some shortcomings (which will be adressed below). A straightforward way of implementation would be to modify at the margin the current allocation methodology by using the actual production instead of the historical activity level multiplied by the activity level threshold. In practice, actual production can only be known one or two years after the year for which the allocation is needed. A prior estimation of production would have to be made with the latest verified data, followed by a correction one or two years later once the production is verified. Such recalculation is not a serious impediment as it is a common practice in other fields, such as taxes or energy and water bills.

This marginal change would leave unfixed other problems from the current allocation methodology. First, the carbon leakage list is exclusive: a sector is either in or out of the list. This approach has generated accrued lobbying from sectors to be on the list, and resulted in a levelling-down of the rules of inclusion.[38] Second, indicators for the inclusion on the carbon leakage list for 2012-2020 are based on a quantitative analysis that seems obsolete. Carbon costs are computed with the assumption of a carbon price of 30 euros per ton (which was based on a forecast

[38] In phase III, 162 sectors were on the list (out of 258), representing 95% of industrial emissions (Ecofys et al., 2013). The third criteria of inclusion (trade intensity superior to 30% regardless of carbon costs) was the most controversial as it is a poor indicator of leakage risk (Clò, 2010; Martin et al., 2014). It added 117 sectors, including "manufacture of wines", "manufacture of weapon and ammunition", "manufacture of perfumes and toilet preparation", "manufacture of jewelry and related articles", "striking of coins" or "manufacture of musical instruments" (European Commission, 2009).

of the EU ETS market in 2008), and with sectors assumed to buy 75% of their allowances.[39] (Juergens et al., 2013). Since 2012, the carbon price has never exceeded 10 euros, and in the cement sector, average unit emissions are on average 10% higher than the benchmark in 2012[40]. With these modifications, computed carbon costs of 60% with the first analysis would shrink to 2.5%.

Based on these two shortcomings, we recommend two guiding principles in the carbon leakage provision. First, the auctioning factor could be (i) *continuous* instead of discrete (in or out), and (ii) dynamic and corrected *ex post* (like production). Data used to compute it would be updated as often as possible depending on administrative costs (every year for the carbon price, but every three or four years for data on gross value added, international trade or unit emissions). The detailed elaboration of such a policy are beyond the scope of this thesis.[41]

Output-based allocation presents some drawbacks, reviewed in Quirion (2009). First, incentive problems can appear with intermediary products. One prominent issue with OBA applied to the cement sector is the so-called "clinker dilemma" (Demailly and Quirion, 2006), whether the considered output is cement or clinker. We recall that almost all emissions in the cement sector are due to the manufacturing of clinker. If allowances are distributed in proportion of cement production, there is an incentive for the producer to import clinker, and sell allowances corresponding to saved emissions. Conversely, if allowances are distributed in proportion of clinker production (clinker benchmark), the incentive to reduce the share of clinker in cement, which has been the main driver of abatement in the cement industry (see Chapter 4), is neutralized. In this specific case, a hybrid benchmark,

[39]Production is implicitly supposed to be equal to the historical activity level in the computation of the European Commission. In practice, production was much lower and many sectors such as cement faced negative carbon costs when taken into account the amount of free allocation.

[40]0.841kg CO_2 per ton of clinker , compared to 0.766 (source: GNR).

[41]We only give a draft of what such rules could look like in this footnote. Another factor would be added to the formula, the free allocation factor, *FAF*, which would be computed such as (i) *FAF* would be comprised between 0 and 1 (ii) if possible with $FAF \leq 1$, carbon costs would not exceed a certain threshold CC_{max}, the latter decreasing with trade intensity (as it would be harder to pass these additional costs to consumers). In such a system, sectors with high *TI* but very low *CC* would have an auctioning factor close to 0. For example, let us choose CC_{max} to be equal to 10% if *TI*=0%, 5% if *TI*>15%, and linear in-between. In the case of the cement sector, we compute with Eurostat data a trade intensity of *TI*=10% (so CC_{max}=7.5%). Exploiting data from the previous European Commission assessment, we find that a carbon price of 10 euros (respectively 5 euros and 20 euros) would lead to an free allocation factor of 0.8 (respectively 0.5 and 0.95).

where distributed allowances depend also on the clinker-to-cement ratio, could solve this dilemma (Branger and Sato, 2015).

Second, economic modeling has shown that OBA entails a higher overall economic cost because it gives too little incentive to reduce the production of polluting goods. In such a system, emission reductions come essentially from the reduction of emissions intensity, not from the output level. To ensure consumption efficiency through the recovery of carbon costs to consumers, some authors (Acworth et al., 2015) have proposed to add to the EU ETS a consumption-based charge.[42]

Under OBA, indirect emissions could also be taken into account for electricity-intensive industries. The same methodology would be used, but a benchmark including indirect emissions would be introduced in place of the benchmark using direct emissions, combined with an electricity emissions factor to be determined (Ecofys, 2014). This would make obsolete the current legislation in which financial compensation for indirect emissions are left to Member States' discretion.[43] Another issue is to make OBA compatible with a fixed cap declining over time. The variations of dynamic free allocation could be immediately compensated by an adjustment of the number allowances auctioned, or as Ecofys (2014) proposes, managed by an ambition-neutral Allocation Supply Reserve. In order to provide a signal towards the long term scarcity of allowances, the California ETS, which is based on OBA, includes a "cap adjustment factor" declining over time. Ecofys (2014) rather suggests differentiated declining benchmarks based on technology roadmaps.

OBA offers a desirable solution for the short and medium term, but a challenge remains in the long term to close the emissions gap between what technology roadmaps enable, and the emissions trajectories needed to avoid detrimental climate change. Providing free allocation may not promote radical innovation, and the ability of the EU ETS to trigger the necessary research efforts to deploy

[42] The charge, based on the EU ETS permit price and the allocation benchmark, would be levied when carbon intensive goods are released for consumption. Such a scheme would be inspired from excise duties on alcohol and tobacco, except that the levy would be assigned to national trusts, separated from national budgets in order to avoid the charge to constitute a tax requiring an unanimous vote in the Council. As all products would be treated equally regardless of origin, the measure would be WTO-compatible.

[43] http://www.emissions-euets.com/carbonleakage

breakthrough technologies has been questionable. In 2004, following the momentum of the EU ETS creation, the European Ultra Low Carbon Steel Making (ULCOS) consortium was launched . Regrouping all major steel companies in Europe and research institutes, it aimed at investigating ways to cut carbon emissions by at least 50%, with a total budget of 75 million euros for the period 2004-2010 (Neuhoff et al., 2014a). With a tenfold estimated budget, the upscaling of the project was finally abandoned because of the low carbon price and the lack of will from the steel industry. In the meantime, an examination of EUTL data[44] reveals that the allowances surplus of the steel sector amounted to 100 million EUAs in phase I and 592 million EUAs in phase II (including 148 million EUAs from offset credits). Making the reasonable assumption that most of this surplus was not earned by technological progress but came from overallocation like in the cement sector, a low bound estimation of the carbon rent obtained from the EU ETS comes to 1.5 billion euros for phase I and 5 billion euros for phase II. It seems clear that only a minuscule part of overallocation profits, if any, was routed towards innovation in low-carbon production processes.

The EU ETS must then be completed with ambitious policies correcting the innovation market failure (Jaffe et al., 2005; Fischer and Newell, 2008). A critical phase is to bring research programs and demonstration projects to industrial viability, referred to as the "valley of death" in management literature (Weyant, 2011). The NER300 program (basically routing the revenues from the sale of 300 million allowances for subsidizing installations of innovative renewable energy technology and CCS) goes towards a good direction. But important challenges lay ahead in terms of governance to ensure that these funds allow the deployment of breakthrough low-carbon technologies in the long run. In addition, a price floor rising over time, as in the California ETS, would help investors secure the viability of low-carbon projects.

A new literature is emerging on the benefits of implementing emissions pric-

[44]We established a plant by plant database of the steel sector (544 installations, including 35 installations suspected to use recycled blast furnace waste gases), but a thorough examination like in Chapter 4 was impaired by the complexity of the sector (different products such as coke, sinter and hot metal that are sometimes integrated in one installation) and the lack of GNR-type data. The computed estimation of allowances surplus depends at the margin of hypotheses regarding waste gases power installations.

ing even in the absence of international binding climate agreements (Edenhofer et al., 2015). These include internalization of national emissions to avoid climate change damages (IPCC, 2014), co-benefits such as air quality and energy security (Parry et al., 2014), building new competitive industries for the "green race" (Fankhauser et al., 2013) and public finance opportunities (Siegmeier et al., 2015). Further research is needed in that direction, because the initiatives of countries will help to close the "emissions gap" between current emissions and trajectories that limit the global temperature increase of more than 2 °C. In this decentralized climate regime, smart policies will be needed to address carbon leakage and competitiveness in order to avoid a levelling down of countries' climate ambitions.

In addition to the issues linked with the implementation of BCAs or OBA that were discussed earlier, several topics deserve a particular attention for future research. First, little is known about investment leakage or changes in production capacities due to climate policies. The challenge is to find appropriate investment data and econometric techniques to single out the effect of climate policies from the many other factors influencing the construction or shutting down of industrial facilities, such as prospects of future demand or energy prices.

Moreover, the field of "negative" leakage induced by technological spillovers, which is only emerging in models (Gerlagh and Kuik, 2014), needs to be developed. Many studies have explored induced innovation and the diffusion of environmental technologies (Popp et al., 2009), but additional research is needed to quantify this effect in terms of avoided emissions in order to close the gap with the carbon leakage literature.

Furthermore, the evaluation of the capacities of different sectors to pass on carbon costs to consumers is key in determining potential adverse effects on profitability. This evaluation is made difficult by the challenges of micro data collection and requirements for advanced econometric techniques. Alexeeva-Talebi (2010, 2011), Oberndorfer et al. (2010) and De Bruyn et al. (2010) find preliminary evidence that industrial sectors are able to pass through additional costs to consumers, but further research is needed to provide more robust conclusions. The increasing carbon constraint in Europe will potentially make pass-through more apparent.

Finally, under the pledge and review approach of the future climate regime, the comparison of climate policies will be a great task to achieve. Metrics assessing climate change mitigation efforts and effectiveness remain basic and unsatisfactory (Aldy and Pizer, 2011), despite significant improvements on data availability regarding domestic policies.[45] Progress in this field is a prerequisite for building border adjustments, linking carbon markets (Ranson and Stavins, 2014), or simply learning which policy designs are the most efficient. In the end, a polycentric climate regime (Ostrom, 2010) gives the opportunity of learning and experimentation, allowing the improvement of the design and implementation of future commitments.

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Colophon

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