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# Essays on climate policy, trade and competitiveness : three essays in applied environmental economics

Oliver Sartor

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THÈSE DE DOCTORAT  
Université Paris I Panthéon-Sorbonne

Mention: **Économie**

Présentée par

**Oliver Sartor**

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Pour obtenir le grade de  
Docteur de l'Université Paris I Panthéon-Sorbonne

**Essays on Climate Policy, Trade and the Competitiveness**

Three essays in applied environmental economics

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**“L’Université Paris 1 - Panthéon Sorbonne n’entend donner aucune approbation, ni improbation aux opinions émises dans cette thèse; elles doivent être considérées comme propres à leur auteur.”**



# Abstract

## English

This thesis focuses on the nexus between climate change policy, international trade and competitiveness for energy-intensive trade-exposed industries (EITEs). In particular, it explores the question: Can climate policy be designed such that policy makers can do away with the concern that any serious attempt to decarbonise EITE sectors will lead to perverse results of offshoring of EITE production and emissions – a phenomenon known as “carbon leakage”? The thesis approaches this question by drawing in particular on the 10 years of experience of the EU with carbon pricing as the dominant tool for decarbonising EITE sectors. This is done in two steps. Firstly, by empirically evaluating existing EU policy solutions and asking whether the policies ultimately meet basic criteria for environmental effectiveness, economic efficiency, and policy coherence with respect to the long-term goals of EU climate mitigation. A number of important policy gaps and incoherencies are identified in this way that are of themselves interesting and of relevant to current policy settings. Secondly, this thesis takes a step back and questions and ultimately challenges the idea that the overarching policy framework of the EU – in particular the dominant role of the EU ETS carbon market – is sufficient for decarbonising these sectors in the longer term.

## Français

Cette thèse se concentre sur le lien entre la politique sur le changement climatique, le commerce international et la compétitivité pour les industries exposées au commerce à forte intensité énergétique (EITE). En particulier, elle explore la question: la politique climatique peut-elle être conçue de telle sorte que les décideurs politiques puissent dissiper l'inquiétude que toute tentative sérieuse de décarbonation des secteurs EITE conduise à des résultats pervers de délocalisation de la production et des émissions EITE (les « fuites de carbone »)? La thèse aborde cette question en s'appuyant notamment sur les dix années d'expérience de l'UE en matière de tarification du carbone en tant qu'instrument dominant de décarbonation des secteurs EITEs. Ceci est fait en deux étapes. Premièrement, en évaluant empiriquement les solutions politiques existantes de l'UE et en se demandant si les politiques répondent finalement aux critères fondamentaux d'efficacité environnementale, d'efficacité économique et de cohérence des politiques par rapport aux objectifs à long terme de l'atténuation climatique de l'UE. Un certain nombre de lacunes et d'incohérences importantes dans les politiques sont identifiées de cette manière, elles-mêmes intéressantes et pertinentes par rapport aux paramètres politiques actuels. Deuxièmement, cette thèse prend du recul et remet en cause l'idée que le cadre politique global de l'UE - en particulier le rôle dominant du marché du carbone EU ETS - est suffisant pour décarboner ces secteurs à plus long terme.

Key words : EU ETS, Emissions Trading, CO<sub>2</sub>, climate change, energy-intensive industries, steel, cement, climate policy, carbon pricing, benchmarking.

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Any PhD Candidate will tell you that a doctoral thesis is a solitary task. This is all the more reason why any support that is offered – matter how seemingly small or insignificant to the person who offers it – is especially meaningful and appreciated. In my case, I would like to acknowledge a few people in particular who provided crucial help and without whom my thesis would not have been finished. These people are: Prof. Mathieu Crozet, for taking me on as a PhD candidate; Prof. Mouez Fodha, for taking over as my Supervisor after Prof. Crozet moved overseas; my father, Frank Sartor, for his continued encouragement that helped me get “over the finish line”; my IDDRI colleague, Henri Waisman, for his pragmatic advice and encouragement; and, most of all, my wife, Florence, for her eternal patience and moral support throughout, from start to finish. I also wish to acknowledge my co-authors on the paper on free allocation and activity levels, namely Misato Sato, Jean Pierre Ponsard, Fred Branger, for their contribution, and, last but not least, the helpful comments provided my thesis jury, which have helped to make my research better.

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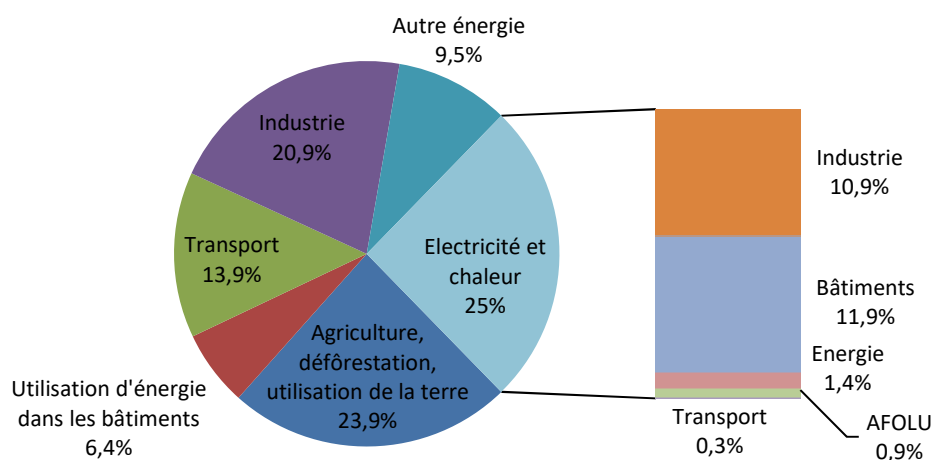
# Synthèse en français

## Le contexte de la politique climatique dans lequel cette thèse a été développé

La motivation pour écrire une thèse sur le thème « Politique du climat, le commerce international et la compétitivité » est née de quelques observations simultanées. La première observation concerne les efforts pour maintenir la température moyenne mondiale en dessous de 2 ° C - le seuil communément utilisé par le Groupe d'experts intergouvernemental sur l'évolution du climat pour décrire les changements climatiques « dangereux » et l'objectif mentionné dans tous les traités internationaux - nécessité une décarbonation importante et profonde de l'industrie lourde. Les procédés de production industrielle représentent environ 21% des émissions mondiales de gaz à effet de serre et 11% des émissions indirectes (en raison de l'utilisation de l'électricité et de la chaleur), soit 32% du total mondial lorsque ces deux chiffres sont ajoutés (voir la figure 1.1).

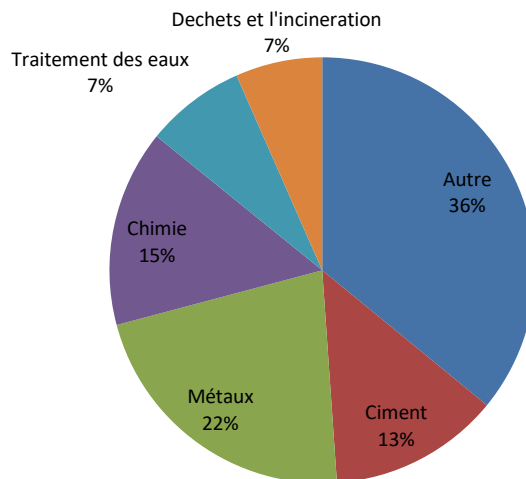
La plupart de ces émissions industrielles proviennent d'une poignée de sous-secteurs très énergivores, les principaux sous-secteurs étant le ciment (13%), les métaux ferreux et non ferreux (principalement le fer, l'acier, l'aluminium et le cuivre (22%), et production chimique (15%), ainsi que quelques secteurs plus petits tels que les productions de minéraux non métalliques (typiquement le verre, la céramique, le plâtre et la brique), les engrais azotés, les pâtes et papiers, le bois, le caoutchouc, la production textile, etc. (voir la figure 1.2).

**Figure 0.1 Émissions industrielles en pourcentage des émissions mondiales totales**



Source: GIEC 2010, [http://ar5-syr.ipcc.ch/topic\\_observedchanges.php](http://ar5-syr.ipcc.ch/topic_observedchanges.php)

**Figure 0.2 Répartition des émissions mondiales directes de GES par industrie**



Source: GIEC, 2010, WG3 AR5 Ch 10 [https://www.ipcc.ch/pdf/assessment-report/ar5/wg3/ipcc\\_wg3\\_ar5\\_chapter10.pdf](https://www.ipcc.ch/pdf/assessment-report/ar5/wg3/ipcc_wg3_ar5_chapter10.pdf)

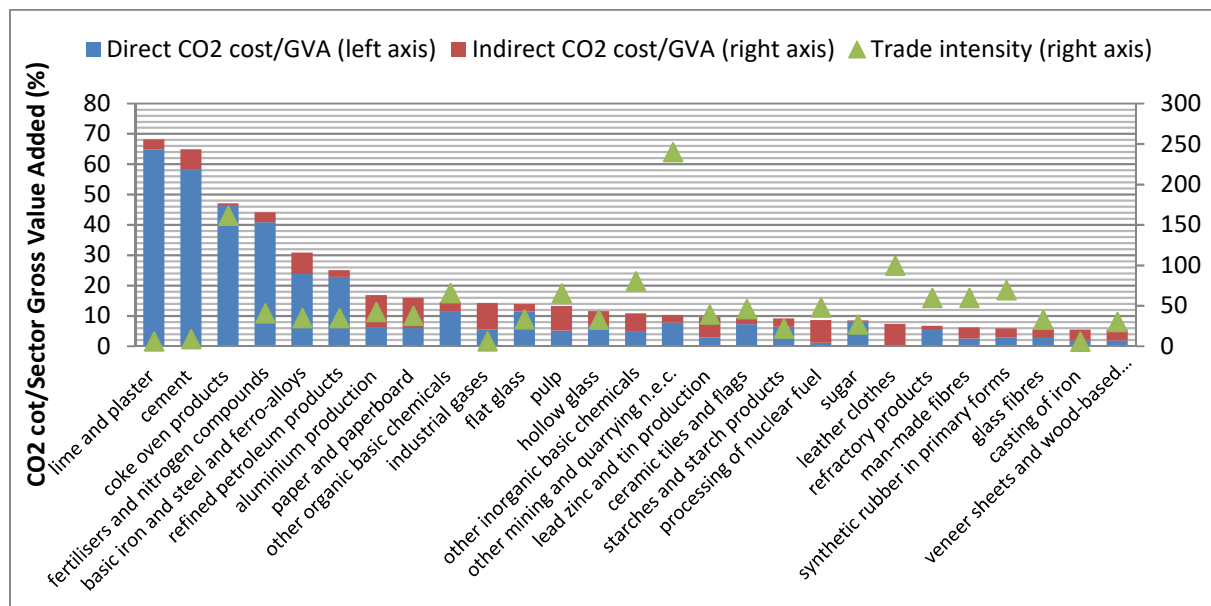
Le Groupe d'experts intergouvernemental sur l'évolution du climat a longtemps suggéré que, au moins pour les pays développés en tant que groupe, il faudrait réduire les émissions nettes de GES de -80 à -95% d'ici 2050 et milieu de la seconde moitié du siècle (GIEC, AR3, 2001 Résumé). Compte tenu des potentiels de réduction dans d'autres secteurs, cela signifie effectivement une décarbonation presque complète de l'utilisation de l'énergie et des processus de production pour les industries énergivores comme le ciment, l'acier et les produits chimiques (voir GIEC AR4 WGIII Ch10). Par conséquent, la nécessité d'une stratégie efficace, rentable et politiquement acceptable pour lutter contre les émissions de l'industrie lourde était (et reste) un élément clé des efforts visant à lutter contre le changement climatique.

Une deuxième observation connexe était que l'industrie à forte consommation d'énergie tend à produire des produits qui sont ou peuvent être échangés sur les marchés internationaux. La combinaison d'une forte intensité d'émission et d'échanges transfrontaliers signifie que les coûts de production induits par une réglementation visant à décarboner la production pourraient mener à des résultats pervers - notamment délocalisation de la production, des investissements, de l'emploi et des émissions dans les régions les moins réglementées. Bien que parfois exagérée par l'industrie quand elle s'oppose à la réglementation, ou par les négationnistes qui prétendent que cette action est trop coûteuse, pour un petit nombre de secteurs énergivores, ce risque - connu dans le jargon comme « les fuites de carbone » - est réel.

La nature de ce risque peut être vue à la figure 1.3. La figure utilise les données de l'industrie et du commerce internationale de l'UE. Il suppose ensuite un prix du CO<sub>2</sub> de 30 € / tonne et calcule ce que le coût supplémentaire pour les industries à forte consommation d'énergie serait en proportion de leur marge brute ajoutée. Les barres bleues représentent le coût direct des émissions, tandis que les barres rouges représentent le coût indirect des émissions (provenant de la consommation d'électricité). Cela donne une indication de la sensibilité des industries aux coûts d'une régulation supplémentaire du CO<sub>2</sub>. La série de triangles verts montre pour chaque industrie une mesure de « l'exposition à la commerce

internationale ». Cet indicateur est mesuré comme le ratio des importations plus les exportations à la production nationale plus les importations (entre les pays de l'UE et les pays hors UE dans ce cas). Il donne une idée de la facilité avec laquelle les produits de l'industrie peuvent être échangés à l'échelle internationale et constitue donc une mesure indicative (même imparfaite) du potentiel de perte de compétitivité d'une production délocalisée. Les résultats montrent qu'il existe une poignée d'environ 12-15 secteurs avec des coûts de CO2 relativement élevés en proportion de la valeur ajoutée à 30 €/tonne. En même temps, la plupart de ces secteurs ont un niveau de commerce international non négligeable. Une exception à cela est la production de ciment et de chaux et de plâtre. Cependant, il convient de noter que dans ce cas, le manque de commerce international reflète un poids élevé sur les ratios de valeur ajoutée, ce qui induit des coûts de transport élevés. Une analyse a cependant été faite qui suggère qu'à des prix de CO2 suffisamment élevés (environ 25 € ou plus) le transport par bateau de l'étranger vers les régions côtières peut devenir économique. Ainsi, le risque de fuite de carbone à des prix significatifs du carbone de 30 €/tCO2 ou plus - c'est-à-dire des niveaux de prix souvent considérés comme modestes par rapport aux prix nécessaires pour conduire une décarbonation importante - peut devenir significatif.

**Figure 0.3 Qui est une industrie exposée à une forte consommation d'énergie?**



En outre, les pays du monde entier continueront très probablement à poursuivre leurs objectifs nationaux de décarbonation à différentes vitesses et de différentes manières. Ainsi, le potentiel de différentiels de coûts de production marginaux significatifs et durables qui affectent les flux commerciaux pour les industries dites « à forte intensité de commerce et à forte intensité énergétique » (EITE en anglais) entre les nations est également réel.

Il était donc clair que les efforts visant à décarboner les industries exposées aux fortes consommations d'énergie devraient trouver des solutions innovantes et économiquement intelligentes au problème de la décarbonation des industries à forte intensité énergétique face à la menace de fuite de carbone. Ces solutions devaient éliminer efficacement les risques de fuite tout en maintenant les incitations à décarboner, être aussi économiquement efficaces que possible et largement acceptables sur le plan politique tant pour l'industrie que pour les groupes de la société civile opposés à la générosité excessive de l'industrie. Cette dimension mondiale différencie fortement l'industrie énergivore d'une perspective stratégique de nombreux autres grands secteurs émetteurs, tels que la production d'électricité, le

chauffage et le refroidissement dans les bâtiments et le transport routier, où les obstacles au commerce transfrontalier sont généralement prohibitifs.

Une quatrième et dernière observation importante était que la plupart des experts et des économistes de l'environnement considéraient les instruments dits « d'échange de droits d'émission » à la fin des années 2000 et au début de 2010 comme le principal vecteur de cette décarbonation. tel que la production d'énergie. Par exemple, l'OCDE, qui reflète généralement la sagesse politique conventionnelle des pays développés, a déclaré que "une utilisation plus large des systèmes d'échange de droits d'émission (ou de la fiscalité environnementale) serait l'un des moyens les plus efficaces de promouvoir la croissance verte". (OECD.org, 2013). Dans ce contexte, l'UE a développé et lancé le système européen d'échange de quotas d'émissions (EU ETS) en 2005 et de nombreuses autres juridictions nationales et infranationales - telles que l'Australie, la Nouvelle-Zélande, le Japon, la Corée du Sud et la Chine. États américains, y compris la Californie et New York, développaient leur propre ETS à la fin des années 2000 et au début des années 2010. Dans ce contexte, l'EU ETS était considéré par beaucoup comme étant non seulement un pilier principal des efforts pour lutter contre le changement climatique dans l'UE, mais également considéré comme un terrain d'essai vital pour l'avenir de la politique climatique au niveau mondial.

Cependant, dès le début de l'EU ETS et au moment de la conception de cette thèse, il était évident que, pour réussir, l'EU ETS - et d'autres systèmes d'échange de quotas d'émissions similaires - devraient trouver une solution satisfaisante et durable à la question du risque des fuites de carbone. Sinon, ces instruments auraient tendance à mourir lentement en raison de leur incapacité à obtenir un soutien suffisamment large parmi les principaux groupes de parties prenantes.

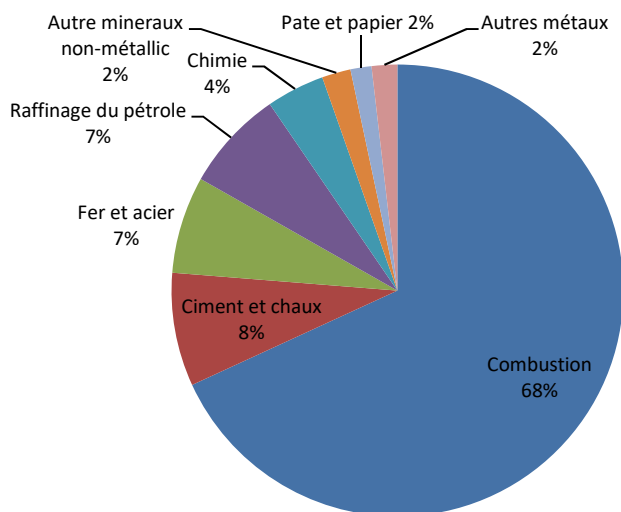
En effet, les préoccupations concernant les impacts négatifs sur la compétitivité internationale et les fuites de carbone ont été l'un des principaux arguments avancés - de bonne foi ou de mauvaise foi - pour justifier la résistance à une politique climatique plus ambitieuse. Un bon exemple de cela est l'EU ETS lui-même. Même après 5 à 7 ans d'expérience avec le système d'échange de quotas d'émission de l'UE, la mise en place d'un système d'échange de droits d'émission efficace s'est avérée difficile. Cela a été difficile en partie à cause de l'opposition des États membres aux secteurs de l'énergie à forte intensité de charbon, comme la Pologne, où environ 90% de l'électricité est encore produite à partir de houille et de lignite. Cependant, au moins autant d'opposition est venue de l'industrie exposée au commerce intensif d'énergie. Pour citer un exemple parmi d'autres, en 2005, lors du lancement de l'ETS, l'association européenne du ciment, Cembureau, affirmait que « l'initiative unilatérale de l'UE de lancer son ETS entraîne des problèmes majeurs de compétitivité de l'industrie cimentière européenne et ne parviendra pas à réduire les émissions mondiales de CO<sub>2</sub> » (Skjaereth et Eikeland, 2016). Depuis lors, le secteur du ciment s'est opposé à plusieurs reprises aux efforts visant à renforcer les incitations créées par EU ETS pour décarboner son secteur, souvent la base des problèmes de compétitivité. Cela comprenait une opposition aux efforts répétés pour résoudre les problèmes importants qui ont surgi dans le contexte du SCEQE, comme par exemple un très important excédent structurel de quotas d'émissions, qui a pesé sur les prix depuis la crise économique mondiale et la crise de la dette de la zone euro. 2009 et 2012 (carte d'influence, 2017). D'autres secteurs EITE ont généralement adopté des positions similaires. Ainsi, le problème de savoir comment trouver les bonnes solutions politiques qui feraient disparaître l'argument de fuite de carbone contre l'ambition dans le contexte des systèmes d'échange de droits d'émission semblait être d'une importance primordiale en tant que question de recherche politique au moment de la conception de cette thèse.

### Encadré 1. L'EU ETS et sa pertinence pour la politique climatique européenne et mondiale

Le système d'échange de quotas d'émission de l'UE est un système multinational de limitation et d'élimination progressive des gaz à effet de serre en Europe. Il s'agit du premier système de « cap and trade » à grande échelle pour les gaz à effet de serre à être mis en place partout dans le monde. En tant que tel, il a suscité un vif intérêt de la part des chercheurs en politique climatique. Il opère dans les États membres de l'UE28 et comprend 3 membres supplémentaires de l'Espace économique européen (Norvège, Islande et Liechtenstein). Les installations d'une capacité thermique nominale de 25 MW ou plus sont généralement tenues de participer au projet. Les principales activités couvertes par le régime sont la combustion pour la production d'électricité et les réseaux de chauffage urbain, les procédés de production industrielle à forte intensité énergétique tels que la production de clinker, les métaux ferreux et non ferreux, le raffinage du pétrole et la production de certains produits chimiques : comme la production de verre, de pâte et de papier, de brique et de céramique. À partir de 2012, les émissions attribuées aux vols intra-EEE par les compagnies aériennes ont également été incluses dans le programme. Le système couvre actuellement environ 12 000 installations de sources ponctuelles dans l'UE.

Le programme est organisé en « phases de conformité » : phase 1 ou phase d'essai (2005-2007), phase 2 (2008-2012), phase 3 (2013-2020), phase 4 (2021-2028). En tant que premier projet pilote mondial, chaque phase a été accompagnée des leçons apprises et d'un cycle de réformes ultérieures visant à améliorer le fonctionnement du système dans les phases ultérieures.

**Figure 0.4 Composition sectorielle des installations ETS par activité principale**



Source: EEA, 2016 Trends and projections in EU ETS

Note: La catégorie « combustion » concerne principalement les installations de production d'électricité et de chaleur pour le secteur résidentiel et le secteur tertiaire. Cependant, il comprend également les activités de combustion liées à la production d'autres catégories dans le graphique ci-dessus, telles que le ciment et la chaux,

le fer et l'acier et les produits chimiques. Ainsi, les chiffres ci-dessus ont tendance à sous-estimer la taille de ces catégories plus petites.

Le système fonctionne en plaçant une limite absolue ou un « plafond » sur les émissions combinées globales des installations couvertes et en délivrant des permis d'émission ou des «quotas de l'Union européenne» équivalents à cette limite chaque année. Après chaque année civile, les entreprises ayant des installations couvertes sont légalement tenues de soumettre un nombre équivalent de quotas correspondant à leurs émissions vérifiées de l'année précédente. Le fait de ne pas soumettre le nombre correct de quotas peut entraîner une amende de 100 € par allocation manquante et une obligation de combler le déficit de l'année suivante. Le principe économique sous-tendant le régime est qu'en émettant progressivement un nombre réduit de quotas aux entités couvertes chaque année, une pénurie de droits d'émission est créée. Les entreprises sont libres d'échanger des quotas entre elles en fonction de leurs besoins de conformité. La combinaison de la rareté et de la négociabilité des quotas permet l'émergence d'un prix du carbone sur un « marché du carbone ». Ce prix permet à son tour aux entreprises d'optimiser leurs décisions d'abattement à court terme, en choisissant d'acheter des quotas ou de réduire et de vendre des quotas en fonction de leurs coûts marginaux d'abattement marginal.

La figure 1.5 ci-dessous montre le prix du carbone de l'EUA entre 2005 et le début du programme et début 2017. On peut voir que le prix a fluctué de manière significative. Ces fluctuations de prix ont été dues à quelques facteurs. La forte baisse et la reprise du prix au cours de la Phase I du Régime étaient dues au caractère expérimental du régime à ce moment-là. En raison de l'incertitude des données sur les émissions historiques réelles avant la création du système, l'UE a été contrainte de s'appuyer sur des estimations peu fiables des émissions de plusieurs pays de l'UE. Cela a conduit à une surabondance de quotas pendant la phase d'essai et, comme les quotas ne pouvaient pas être stockés dans les phases futures, l'équilibre des prix après une certaine réduction s'est établi à zéro. Dans la phase II du régime, le prix a recommencé à environ 28 € / tonne de CO<sub>2</sub>, pour tomber une seule fois suite à l'effondrement de la production industrielle et énergétique suite aux crises macro-économiques de cette période. Cela a également été exacerbé par les dispositions de la conception du régime qui ont permis d'ajouter un grand nombre de crédits compensatoires internationaux à l'offre existante excédentaire de quotas. Depuis lors, l'amélioration continue de l'efficacité énergétique, l'augmentation de la part des énergies renouvelables, la faiblesse de la production industrielle et la fermeture de surcapacités dans plusieurs secteurs industriels à forte intensité énergétique ont contribué à une surabondance importante et durable des quotas. et très peu d'incitations à la réduction.

**Figure 0.5 Évolution du prix des quotas de CO<sub>2</sub> EU ETS (de janvier 2005 à juillet 2017)**



## Résultats principaux issus des questions de recherche abordées

La recherche contenue dans les chapitres suivants a révélé un certain nombre d'idées intéressantes. Ces idées devraient être considérées comme faisant partie de la littérature sur les politiques générales, et en particulier de la littérature sur l'analyse des politiques ex post, sur les systèmes d'échange de droits d'émission. Ainsi, l'accent est fortement axé sur les politiques et conçu pour être consommable, dans la mesure du possible pour une thèse de doctorat, par des experts politiques et des décideurs politiques. En conséquence, l'approche méthodologique a cherché à trouver un équilibre entre la rigueur intellectuelle et économique et la lisibilité des personnes moins expertes dans le domaine de la politique climatique.

Chacun des trois chapitres suivants aborde une question distincte et a été initialement rédigé pour une publication indépendante. Cependant, les trois chapitres sont reliés par un objectif thématique commun. Plus précisément, chacune de ces questions vise à répondre au problème de la façon de combiner des incitations efficaces pour la décarbonation de l'industrie exposée à une forte consommation d'énergie tout en minimisant le risque de fuite de carbone.

Le chapitre 2 se penche spécifiquement sur l'introduction d'une nouvelle réforme des secteurs exposés aux fuites de carbone dans l'EU ETS en 2012/2013, sous la forme de quotas gratuits basés sur des « benchmarks » ou standards de performance des émissions. La question des points de référence a immédiatement été intéressante pour mon sujet, car il s'agissait d'une tentative de l'UE de répondre à un ensemble de critiques et de plaintes déposées par différents groupes de parties prenantes au cours de la phase de test ETS. Ces critiques portaient sur la façon dont les risques liés à l'industrie et aux fuites de carbone, qui consomment beaucoup d'énergie, étaient gérés par la conception même de l'ETS. Parmi les sujets de préoccupation figuraient le fait que l'approche flexible consistant à laisser les États membres choisir leurs systèmes d'allocation gratuits conduisait à des transferts massifs de prestations sociales vers certaines installations ou industries; que le marché intérieur était potentiellement faussé; et que les incitations à la réduction étaient diluées.

Pour répondre à ces préoccupations, l'UE a donc réagi en mettant en place un système d'allocations gratuites aux secteurs exposés aux fuites de carbone sur la base des meilleurs repères technologiques prédéterminés disponibles. La prémisse était que seulement le top 10% des installations dans un secteur donné recevrait 100% de leurs quotas gratuitement, créant ainsi une cible pour les installations à la traîne à viser. À l'époque, c'était la première fois qu'une telle analyse comparative avait été utilisée pour l'industrie dans la politique climatique.

Ce chapitre a été écrit précisément au moment de la publication des premières données sur le nouveau mécanisme (en 2012). Mes co-auteurs et moi-même avons donc cherché à utiliser ces nouvelles données détaillées et de qualité relativement élevée sur des installations ETS individuelles pour analyser ce que le nouveau système signifiait. Plus précisément, nous avons cherché à être les premiers à utiliser ces données pour comparer les règles d'allocation dans les phases 1 et 2 par rapport à la phase 3 et leurs implications. Compte tenu du coût et de la complexité du développement et de l'administration du nouveau mécanisme (~ 25-30 millions €), 3 critères ont été évalués:

- les implications distributionnelles du bien-être du changement dans les règles;
- l'adéquation de la protection contre le risque de fuite de carbone, et



- le risque de distorsions du marché intérieur qui a été éliminé ou qui est resté dû au système.

Nous avons constaté que les nouvelles données publiées et soigneusement collectées et analysées permettaient en effet une évaluation plus précise des impacts des règles d'allocation gratuite dans les Phases 1 et 2 du SEQE-UE et, bien que commençant, des prédictions et des dessinés au sujet de la phase 3.

Plus précisément, nous étions sur le point de conclure que le passage à une analyse comparative harmonisée avait effectivement apporté des avantages importants par rapport au système des droits acquis. Plus particulièrement, il avait conduit à une réduction significative des subventions injustifiées à l'industrie européenne d'une valeur de plusieurs dizaines de milliards d'euros au cours de la seule phase 3, sous les prix du carbone prévus. Le mot injustifié fait référence au fait que ces attributions gratuites dépassaient les émissions totales et qu'elles compensaient donc effectivement les coûts de réglementation nouvellement encourus. En plus de cet avantage, les nouvelles règles ont également introduit plus de transparence dans la méthodologie derrière les allocations gratuites et, couplées à l'utilisation d'un règlement (plus ou moins) commun pour toutes les installations en Europe, cela a créé un système à la fois mais a également été largement considéré comme plus équitable par les parties prenantes (donner ou prendre quelques faiblesses restantes, voir le chapitre 3).

Cependant, l'étude a également mis en évidence certaines faiblesses du nouveau mécanisme. Tout d'abord, nous avons constaté qu'il peut protéger contre les fuites à des prix plus élevés du CO<sub>2</sub> (par exemple au CO<sub>2</sub> supérieur à 25 € / tCO<sub>2</sub>) ou dans un contexte de croissance où les productions historiques (sur lesquelles sont basées les allocations totales) sont dépassées. Le mécanisme était également limité en tant que moteur des innovations technologiques à faible émission de carbone, car il ne prévoyait aucun moyen de payer les coûts supplémentaires de l'innovation sur le prix du carbone. Avec le recul, cela constitue et reste une faiblesse majeure de la structure d'incitation créée pour la décarbonation industrielle dans le cadre de l'approche d'allocation gratuite basée sur l'indice ETS +.

Pour être précis, il y avait aussi quelques faiblesses dues à des aspects de conception spécifiques de la façon dont l'UE a mis en œuvre l'approche de benchmarking et pas simplement l'analyse comparative en tant que telle. Par exemple, l'UE a choisi que les allocations soient basées sur un indice de référence multiplié par des niveaux de production "ex-ante", associés à un ensemble de seuils déclenchant des "récupérations" sur les futurs niveaux d'allocation libres (voir la chapitre 3). En outre, l'UE a compliqué davantage la politique et l'économie de son système d'allocation basé sur des benchmarks en faisant le choix d'appliquer un « facteur de correction trans-sectoriel » aux allocations gratuites pour éviter que la part totale de l'allocation ne croisse le plafond du système sur les émissions. Ces deux choix signifiaient effectivement que les allocations futures, même avec des repères, pourraient et pourraient encore dévier, parfois considérablement, des émissions réelles. En rétrospective, ces décisions se sont révélées très controversées. Ils ont également, de l'avis de l'auteur, porté préjudice à l'objectif de la politique consistant à écarter « l'argument des fuites de carbone » en tant qu'obstacle au renforcement du prix du carbone au sein même du système et donc à une décarbonation plus rapide et plus profonde.

Enfin, le document fournit d'autres idées intéressantes sur la manière dont les États membres de l'UE ont effectivement distribué des quotas gratuits selon des règles plus souples avant l'introduction du système d'analyse comparative. En effet, les États membres n'étaient pas seulement des allocations de droits acquis, comme certains l'ont suggéré dans la littérature (par exemple Ellerman et al, 2010). Plutôt, le « grandfathering » n'était pas simplement le maintien des droits historiques d'émissions, il était des droits historiques de droits acquis plus une allocation supplémentaire d'allocation. Cela peut peut-être

être interprété comme un aperçu de la façon dont les pays participant à un système commun d'échange d'émissions peuvent être incités à agir, lorsque le désir de soutenir les industries nationales coïncide avec la création d'une nouvelle source de valeur (certificats d'émission).

Bien que cette situation ait engendré un gaspillage total pour l'industrie, nous avons néanmoins constaté qu'il était difficile de justifier le coût de la mise en œuvre du nouveau système uniquement en termes de création de conditions plus égales pour les entreprises en Europe. Tandis que cela était vrai, de nombreux États membres avaient tendance à sur-allouer leurs industries par des pourcentages similaires, et comme les coûts commerciaux de nombreux produits concernés (par exemple le clinker) étaient assez élevés par rapport aux coûts implicites de carbone par unité de produit, le l'impact global sur les conditions de concurrence équitables du marché intérieur de l'UE n'était pas un problème majeur qui devait être traité - ou du moins pas à ce stade du système.

Le chapitre 3 de la thèse se penche sur un autre changement important dans la phase 3 de l'EU-ETS pour les EITE, qui concernait l'introduction de la règle du « seuil de niveau d'activité » (« activity level threshold » ou ALT en anglais). La raison sous-jacente de son introduction est qu'elle réduirait les bénéfices de sur-allocation à l'industrie recevant une allocation gratuite en cas de chute de la demande: chaque fois que le niveau d'activité d'une installation tombe en dessous d'un seuil (50%, 25%, 10%) par rapport à son niveau d'activité historique utilisé pour allouer des allocations gratuites, l'allocation serait réduite en conséquence (50%, 25%, 0%). Cependant, la préoccupation de mes co-auteurs et de moi était qu'en créant de tels seuils, de fortes incitations seraient créées pour jouer le système. Nous avons donc décidé d'apporter cette question aux données et de fournir la première évaluation de la règle dans la littérature.

Notre analyse ex post de l'année 2012, première année d'application de la règle du seuil, s'est concentrée sur le secteur du ciment, secteur dans lequel environ la moitié des pays de l'UE avaient connu une baisse significative de la consommation (pays LD). Le secteur du ciment était un bon candidat à examiner de manière empirique, car il présente un lien technologique simple entre les émissions et les niveaux de production, ce qui rend l'utilisation de variables indirectes pour estimer la production plus fiable. Il s'agissait également d'un secteur à très forte intensité de carbone par unité de production, ce qui rend les incitations économiques à jouer le système potentiellement très fort dans certaines circonstances. "Ciment en 2012" a donc fourni une expérience naturelle pour évaluer les conséquences de cette règle.

Notre conclusion principale est que si les ALT ont réduit dans une certaine mesure les bénéfices de sur-allocation, ils ont également créé des distorsions opérationnelles importantes qui ont conduit à des résultats incompatibles avec la transition à faible intensité de carbone des industries à forte intensité énergétique de l'UE. Plus précisément, il a montré qu'il existait des preuves évidentes d'effets de seuil, les entreprises cimentières trouvant des moyens d'augmenter la production de clinker (la partie la plus intensive des émissions de ciment) au-dessus de ce qu'elles auraient autrement produit pour franchir le seuil à 50% et à 25% pour ensuite toucher des quotas supplémentaires dans l'allocation de l'année suivante. Nous avons identifié trois façons dont les activités des entreprises sont modifiées en réponse aux ALT: déplacement de la production entre les usines, augmentation des exportations nettes de clinker et de ciment, augmentation du ratio clinker / ciment.

Une conséquence évidente est également que la réduction des bénéfices de sur-allocation est également moins importante que prévu en raison du comportement de jeu de l'industrie pour atteindre les seuils, pendant les périodes de faible demande du marché.

Les distorsions opérationnelles rapportées dans notre étude sur le déplacement de la production vont à l'encontre de la restructuration des actifs pour réaliser des économies d'échelle, un facteur clé de la rentabilité dans le ciment; l'augmentation des exportations induit une certaine délocalisation de la production étrangère de ciment dans l'UE, générant des inefficacités de coût et des émissions supplémentaires dues au transport; l'augmentation du ratio clinker / ciment va à l'encontre de l'un des principaux moteurs de la limitation des émissions dans la production de ciment. En résumé, les distorsions générées par l'introduction de l'ALT ont entravé l'alignement progressif des mesures d'incitation sur la transformation à faible intensité de carbone dans ce secteur.

La preuve que les effets de seuil causaient des distorsions significatives était très forte même sans analyse contrefactuelle détaillée (voir la figure 1.6 par exemple). Une question plus difficile est: quelle était la taille de la distorsion? Et: dans quelle mesure a-t-il porté atteinte à l'objectif de la Commission, à savoir des allocations excessives limitées aux entreprises qui avaient considérablement réduit leurs activités? Pour répondre à cette question, un scénario contrefactuel a été développé.

Avec l'élaboration d'un contrefactuel, nous avons estimé que dans notre sous-ensemble de pays « à demande faible » de l'UE, la production de clinker avait augmenté d'environ 7,2 mégatonnes en 2012 et que les émissions de ces pays avaient augmenté d'environ 6,4 mégatonnes de CO<sub>2</sub>. Les preuves de telles augmentations dans les pays à forte demande étaient cependant un peu plus faibles, ce qui laisse penser qu'un tel comportement était au mieux marginal (comme on pouvait s'y attendre). Pour mettre ceci en perspective, cela se compare à un marché de clinker d'environ 134 Mt en 2012 et aux émissions totales du secteur du ciment et de l'EU ETS en 2012 d'environ 113 MtCO<sub>2</sub> et 1800 MtCO<sub>2</sub>eq respectivement. Les distorsions, si elles sont correctement estimées, sont significativement significatives pour le secteur du ciment (~ + 6% des émissions / production) et quantitativement importantes en termes absolus, bien qu'elles soient modestes par rapport à la taille des émissions actuellement couvertes par le SCEQE (qui, après tout, couvre environ 45% des émissions totales de CO<sub>2</sub> de l'UE).

En termes quantifiés qu'après l'introduction des ALT: le bénéfice potentiel de sur-allocation pour le secteur du ciment de l'UE avec du comportement pervers est de 278 M € (2 € / t clinker) et de 207 M € sans du comportement pervers, alors qu'il aurait été de 368 M € l'absence d'ALT. Les incitations sont amplifiées dans les pays à « faible demande », où le profit avec du comportement pervers est de 213 M € (3,9 € / t clinker) et de 158 M € sans du comportement pervers, alors qu'il aurait été de 306 M € sans ALT. Cela suggérerait que les efforts de la Commission pour limiter les retombées de l'allocation gratuite à l'industrie du ciment en utilisant au moins cette approche n'ont été que marginalement efficaces, tout en induisant des effets secondaires environnementaux, économiques (et potentiellement politiques) très négatifs.

Enfin, il convient de noter que nos résultats ont été obtenus dans un contexte de faible prix du carbone, de forte baisse de la demande du marché et d'importantes allocations gratuites de quotas. Cependant, un prix du carbone plus élevé rendrait nos résultats encore plus pertinents; plus le prix du carbone est élevé, plus l'incitation à atteindre les seuils est élevée. Certes, si nous avions observé une croissance de la demande, la règle du seuil aurait pu être moins pertinente. Cependant, les effets de seuil peuvent alors avoir été de l'autre côté de l'équation, la réserve pour les nouveaux entrants pouvant avoir été une source plus importante de distorsions (il y aurait une incitation à avoir une production artificiellement élevée pendant la période l'équivalent de HAL pour les nouveaux entrants).

Maintenant, la construction du scénario contre factuel dépend évidemment des données disponibles. Idéalement, on aurait aimé avoir pour chaque installation ETS dans l'échantillon de pays à faible

demande une installation de ciment identique juste à côté qui ne figurait pas dans le EU ETS pour lequel des données de production et d'émissions étaient disponibles. Malheureusement, une telle "analyse d'appariement" n'était pas réalisable en raison d'un manque de cimenterie non-ETS dans les pays concernés et d'un manque de données sur leur production pour les très rares qui existent. Pour les installations que nous avons, nous avons géré un grand nombre de variables pour lesquelles des données étaient disponibles au niveau de l'usine (caractéristiques de localisation, différences avec la moyenne des années précédentes, niveau d'activité de l'année précédente, mannequins d'entreprise, etc.). Le meilleur ajustement possible pour les données observées dans le passé. Au final, nous avons pu arriver à un modèle simple qui explique 85% de la variation des niveaux de production des installations. Même si nos estimations ne sont peut-être pas extrêmement précises, nous étions néanmoins convaincus que les ordres de grandeur reflétaient largement ce qui s'est passé en 2012.

Cette confiance découle en partie de la confiance que nous avons dans le modèle contrefactuel lui-même, ainsi que du fait que plusieurs différentes analyses statistiques que nous fournissons (données commerciales, données au niveau de l'entreprise et de l'analyse statistique brute des chiffres) semblent toutes indiquer des effets de l'ampleur que nous trouvons dans les résultats contrefactuels. Cependant, en plus des preuves quantitatives présentées ici, nous avons interviewé plus tard des cadres du secteur du ciment et des gestionnaires opérationnels qui ont confirmé que ce « jeu » des règles de seuil. Leurs propres commentaires sur nos résultats ne semblent pas avoir beaucoup de doute sur l'existence du comportement de jeu ou les ordres de grandeur approximatifs que nous avons trouvés. Ces considérations suggèrent que les seuils de niveau d'activité pourraient devoir être reconsidérés pour des secteurs tels que le ciment pour lesquels les coûts du carbone représentent une part importante des coûts de production. Cela soulève la question de savoir quoi mettre à sa place à la place. Comme mentionné dans l'introduction, les économistes s'accordent à dire que, en l'absence de prix mondiaux du carbone, le remplacement de l'allocation gratuite par une mise aux enchères complète et l'utilisation des ajustements de carbone frontaliers constitue la solution la plus efficace. En effet, cela aide à niveler les coûts du carbone entre les producteurs nationaux et étrangers tout en permettant que les coûts du carbone soient répercutés le long de la chaîne de valeur pour encourager la réduction de la demande. Politiquement, cette solution n'a pas encore acquis une traction sérieuse. Cela s'explique en grande partie par le fait que le nivellement par les frontières peut être perçu comme un protectionnisme déguisé en écologisme et donc pas propice à la confiance dans les négociations internationales sur le climat. Cependant, la situation peut changer. Si l'on se projette dans la période post-2020, on s'attend à ce qu'un plus grand nombre de pays aient commencé à appliquer les prix du carbone. Davantage de pays seront confrontés à des défis similaires liés à la conception de mesures anti-fuites adéquates auxquelles l'UE est désormais confrontée et il pourrait donc y avoir davantage de possibilités d'approches coopératives. Le nivellement des frontières par la coopération internationale prendrait toutefois du temps à négocier et à concevoir. Cela soulève la question de la solution provisoire.

Une option consiste à augmenter le nombre de seuils de niveau d'activité pour réduire l'incitation à la production de jeux. Par exemple, un seuil de 50%, 60% et 70% pour le ciment peut inciter un plus grand nombre d'installations à augmenter leur production de clinker au seuil supérieur suivant. Puisque les seuils créent un système d'allocation entre un système ex-ante et ex-post, il serait beaucoup plus simple de mettre en place une allocation basée sur la production pour des secteurs tels que le ciment où le risque de distorsion est élevé. L'analyse de ce document suggère que cette option surpasserait l'allocation ex ante avec et sans seuil en termes de réduction des distorsions et des profits sur allocation.

Cependant, nous avons également soutenu qu'un certain nombre de questions doivent être soigneusement examinées avant d'aller dans cette direction. Un inconvénient majeur du passage à l'OBA

est que l'on peut s'attendre à peu de chose en termes de répercussion des prix du carbone sur les prix des produits et, partant, de substitution de la demande aux produits à plus faible teneur en carbone. Pour les secteurs où les coûts du carbone sont élevés en proportion des coûts de production, tels que le ciment, cela limiterait considérablement le potentiel de l'UE à réduire les émissions de manière rentable et à décarboner ces secteurs. Contrairement à l'allocation ex ante, l'ABO implique la perte d'un plafond absolu pour les allocations gratuites, ce qui peut être un point politiquement controversé. En outre, la mise en œuvre de l'OBA dans certains secteurs, mais pas tous, peut également poser des problèmes politiques. Des discussions sont en cours sur la façon de contourner ces problèmes. Par exemple, la perte d'incitations à la substitution du côté de la demande pourrait peut-être être restaurée avec une charge de consommation sur les produits en aval (Neuhoff et al 2014a). Une autre alternative pourrait être d'oublier l'allocation gratuite et de mettre en place un ajustement fiscal à la frontière; cependant, cela se heurte également à des défis politiques et diplomatiques majeurs.

Le chapitre 4 de la thèse se penche sur la question de savoir si les entreprises de SCEQE dans les industries dites à forte intensité énergétique exposées au commerce répercutent le coût en carbone de leur processus de production sur le prix de leurs produits. Le chapitre commence par noter que des études récentes sur les potentiels de réduction des émissions de certains grands secteurs de l'EITE comme le ciment et l'acier démontrent que, compte tenu des technologies actuelles, il est impossible de réduire suffisamment les émissions sauf si les consommateurs réduisent aussi leur consommation des matériaux à forte intensité de carbone (Neuhoff et al, 2015a, Neuhoff et al, 2015b). En bref, nous aurons besoin de:

- une plus grande efficacité dans l'utilisation de ces matériaux par les consommateurs,
- une plus grande substitution entre les matériaux à haute teneur en carbone et à faible teneur en carbone dans les applications appropriées, et
- des incitations économiques plus fortes pour l'innovation de produits qui réduisent l'empreinte carbone des matériaux utilisés.

En principe, c'est un rôle que l'EU ETS devrait jouer. Après tout, il existe également une complexité considérable de la gamme de produits, des applications d'utilisation finale et des substituts potentiels tout au long de la chaîne de valeur dans les secteurs de matériaux à forte intensité de carbone. Cette complexité rend improbable que des niveaux appropriés de substitution de produits, d'efficacité matérielle et de soutien à l'innovation puissent être exigés par un régulateur centralisé d'une manière efficace et politiquement durable. En bref, un prix du carbone robuste qui est répercuté le long de la chaîne de valeur dans les secteurs des matériaux à forte consommation d'énergie semble nécessaire.

Cela soulève à son tour la question de savoir si les prix du carbone sont effectivement répercutés sur les prix des produits par les industries EITE. Ce chapitre note que les preuves sur cette question sont loin d'être complètes. Il note également que la théorie économique de base des coûts d'opportunité suggère que la répercussion des prix est évidente et que la Commission européenne a tendance à utiliser cet argument théorique pour justifier les tentatives de réduction de l'allocation gratuite à l'industrie. D'un autre côté, il note qu'il existe dans le débat des arguments plausibles qui sont parfois avancés sur la raison pour laquelle la théorie de la répercussion des prix, même en présence de quotas d'émission librement attribués, pourrait ne pas être vraie dans la pratique. La question a donc une fois de plus été très pertinente pour la conception de mesures anti-fuites dans le cadre du système d'échange de quotas d'émission de l'UE.

J'ai donc cherché à répondre à cette question en examinant deux secteurs du commerce international dans l'EU ETS qui ont les coûts de CO2 les plus élevés par unité de production, à savoir le ciment et l'acier de base. L'hypothèse implicite est que si les secteurs EITE transmettent leurs coûts de CO2 aux consommateurs, ces deux secteurs sont tellement intensifs en CO2 par unité de produit que les preuves devraient être assez évidentes.

Après beaucoup d'économétrie et d'analyse statistique, le résultat que j'ai trouvé était troublant pour les défenseurs du système actuel de protection anti-fuites dans le cadre de l'EU ETS. En bref, le chapitre a montré qu'en fait, les prix du carbone ETS de l'UE n'ont pas été répercutés sur les prix du ciment et des produits sidérurgiques, ou s'il y a eu une telle marge pratiquement inutilisable en tant que source d'incitation pour les consommateurs réduire leur demande matérielle, ou les remplacer, ou être prêt à payer pour des versions innovantes plus coûteuses des mêmes produits.

Les implications de cette découverte sont très fortes. Cela signifie essentiellement que l'EU ETS, même s'il devait surmonter ses problèmes actuels en termes d'offre excédentaire et de manque de « bancabilité » pour les investisseurs, serait néanmoins inefficace pour conduire la décarbonation de l'industrie à forte intensité énergétique telle qu'elle est actuellement conçue. Pour surmonter cette faiblesse, il faudrait procéder à des réformes importantes également à l'ensemble de l'approche visant à éviter les fuites de carbone. En résumé, l'attribution gratuite ex ante devrait être abandonnée et il faudrait soit un système d'ajustements aux frontières et une mise aux enchères complète, soit une attribution ex post gratuite avec une redevance carbone en aval. C'est déconcertant car cela implique une rupture radicale avec la pensée actuelle dans de grandes parties du monde de la politique climatique.

## **Un contexte politique en évolution**

Depuis le début de ce travail, le débat sur la politique climatique et le contexte politique ont également considérablement changé et, à certains égards, ont été imprévus par l'auteur au début de ce travail. D'une part, une plus grande expérience a été acquise en matière de politique climatique et d'échange des droits d'émission en tant qu'instrument de la politique climatique en Europe et ailleurs. En outre, le contexte global de la politique climatique internationale a pris d'importantes avancées intellectuelles (comme avec l'Accord de Paris) et rétrospectives (comme avec la montée du populisme politique aux États-Unis et en Europe). La montée du populisme politique a fait du commerce international une question beaucoup plus politisée et polarisante, avec des ramifications importantes pour la politique climatique. En outre, à la suite d'une Union européenne plus divisée, il est devenu de plus en plus évident que les institutions de l'UE auront du mal à gouverner la politique paneuropéenne du climat et de l'énergie aussi facilement que les créateurs de l'EU ETS. Dans le domaine du climat et de l'énergie, les discours d'experts sur le soutien aux sous-groupements régionaux de « leaders » et l'habilitation par l'UE de stratégies nationales ou régionales pour s'attaquer à des problèmes politiques épineux sont de plus en plus abordés.

Ces développements sont tous pertinents aux sujets abordés et aux conclusions tirées dans les différents chapitres de cette thèse parce qu'ils remettent en question certaines des hypothèses implicites qui sous-tendent le travail. Par exemple, pour les raisons soulignées ci-dessus, cette thèse met fortement l'accent sur la tarification du carbone et le commerce des émissions, en particulier comme un pilier vital et potentiellement même principal des efforts de décarbonation de l'industrie à forte intensité énergétique. Aujourd'hui, avec le recul, cette hypothèse mérite d'être remise en question beaucoup plus sérieusement.

Une faiblesse importante du système d'échange de quotas d'émission de l'UE qui a été révélée avec l'avantage du temps est qu'il est peu probable qu'il soit capable de fournir des incitations suffisantes pour les investissements dans les technologies à faible intensité de carbone et à forte intensité de capital. Il y a quelques raisons à cela. Le premier est que l'évolution des prix de l'EU ETS a montré que le marché est, en effet, incapable de délivrer un signal de prix pertinent qui reflète la pénurie de carbone à long terme. L'expérience a révélé que les prix reflètent presque entièrement les équilibres à court terme entre l'offre et la demande sur le marché, tandis que les fondamentaux des prix à long terme ont peu ou pas d'impact (Schopp et al, 2012, Sartor et al, 2013). Étant donné que de nombreux investissements dans la technologie de la décarbonation profonde destinés aux industries énergivores ont une durée de vie financière de 20 ans ou plus, l'EU ETS n'envoie clairement pas les signaux de prix nécessaires à l'industrie pour décarboner en temps opportun.

Une autre faiblesse de l'EU ETS (et plus généralement des instruments ETS) est que les financiers exigent généralement d'autres garanties ou justifications politiques pour l'analyse de rentabilisation sous-jacente à tout investissement à faible carbone en plus du prix du carbone EU ETS. Cela semble être dû à une expérience historique limitée des prix sur lequel on pourrait baser de la modélisation sûre, un risque politique élevé perçu qui est difficile à évaluer et, malheureusement, un mauvais bilan de générer des prix du carbone significatif. Ainsi, la plupart des sociétés de financement pour les investissements importants dans la technologie à faible émission de carbone ont encore du mal à attribuer pour intégrer les prix du CO<sub>2</sub> de l'EU ETS dans leurs modèles de prêt. Par conséquent, même si des signaux de pénurie de carbone à long terme pourraient être fournis par le marché pour inciter l'industrie à investir dans des technologies de décarbonation à forte intensité de capital, ils auraient encore du mal à financer le déploiement et la commercialisation de leurs investissements.

En outre, dans le cas de l'Europe, l'expérience de l'EU ETS a démontré l'extrême difficulté de générer et de maintenir le soutien politique nécessaire pour même fournir une pénurie de quotas à un ETS à court terme. Cela a été amplement démontré par la difficulté politique suprême rencontrée dans l'UE pour faire face à un excédent structurel massif de quotas d'émissions généré lors de la Grande Récession. Ainsi, dans la pratique, les prix du carbone sont restés plus ou moins insignifiants pour inciter à la réduction, investir dans des investissements relativement onéreux dans de nouveaux équipements bas carbone (comme le CSC) ou investir dans la commercialisation de matériaux de substitution innovants.

Cette difficulté politique de générer des incitations économiques significatives pour la décarbonation dans un ETS est en partie un problème européen - étant donné le défi de s'entendre entre 27 ou 28 Etats membres différents avec des profils économiques et d'émissions très différents. Cependant, l'UE n'est pas seule dans ce domaine. D'autres pays tels que le Japon, les États progressistes des États-Unis et du Canada, l'Australie et la Nouvelle-Zélande - tous salués comme susceptibles de suivre l'UE dans le club d'échange d'émissions - ont depuis constaté des progrès lents ou inversés. Ce phénomène a plusieurs causes. Cependant, de l'avis de l'auteur, un facteur commun à plusieurs de ces cas est que les systèmes de tarification du carbone et d'échange de droits d'émission ignorent en particulier les préoccupations d'économie politique fondamentale des principales parties prenantes. Plus particulièrement, ils sont généralement mis en œuvre en mettant l'accent sur leur « neutralité technologique » et « optimalité économique pour la société dans son ensemble », mais les décideurs accordent peu d'attention aux changements structurels révolutionnaires qui sont requis des industries pour changer complètement leur technologie, et pour changer leur stock de capital et leur modèles commerciaux. Ainsi, les ETS sont souvent introduits sans un récit d'accompagnement ou une vision de la façon dont les industries à forte consommation d'énergie peuvent se transformer pour survivre et être rentables dans un monde

décarboné. Il n'est donc pas surprenant qu'ils soient perçus par l'industrie comme des mesures punitives qui attaquent directement leur capacité de survivre et de fonctionner financièrement.

De plus, cela se produit dans un contexte où la mondialisation et la Grande Récession ont déjà exercé une énorme pression sur les finances des industries de matières premières à forte intensité énergétique dans de nombreuses parties du monde. Bref, même en tant qu'économiste autoproclamé, l'auteur est de plus en plus convaincu que l'action climatique visant à décarboner les industries à forte consommation d'énergie a été quelque peu entachée par une recherche désespérée de pureté économique - sous forme d'échange de droits d'émission les marchés - et l'incapacité à garder un œil sur les forces sociales et politiques importantes qui façonnent l'économie politique environnante en apportant un changement durable.

Maintenant, cela ne signifie pas que l'auteur estime que l'EU ETS n'est pas pertinent pour la décarbonation de l'industrie. L'EU ETS a encore quelques avantages potentiellement importants pour s'attaquer à la décarbonation industrielle qui ne peuvent pas non plus être ignorés. L'un de ses avantages est de fournir un plafond absolu sur les émissions de GES pour un large éventail de secteurs de l'économie, y compris les industries à forte consommation d'énergie. À long terme, cela devrait devenir de plus en plus important à mesure que le budget total des émissions se rétrécira et que la question d'une concurrence équitable entre les pays et les industries participant au marché unique de l'UE deviendra plus pertinente. Un autre avantage est que l'EU ETS est, au moins en principe, capable de fournir des incitations cohérentes tout au long de la chaîne de valeur de la production à forte intensité énergétique. Il y a, après tout, de nombreux vecteurs pour décarboner l'industrie, de la technologie des processus de production et du changement de combustible, du côté de l'offre, à l'augmentation de l'efficacité de l'utilisation finale et de la substitution du côté de la demande. Comme indiqué au chapitre 4 de cette thèse, il reste difficile de voir comment tous ces vecteurs, en particulier ceux du côté de la demande, pourraient être régulés de manière simple, approfondie et sans distorsion sans l'aide d'instruments de tarification. Bien sûr, à court terme, on pourrait imaginer sacrifier une certaine efficacité et vivre avec certaines distorsions pour des raisons d'efficacité. À long terme, toutefois, de telles approches peuvent également se heurter à leurs limites en matière d'économie politique.

Alors, que signifie cette analyse pour comment interpréter les conclusions dans chacun des chapitres suivants? Premièrement, cela signifie que l'on doit interpréter les parties de l'analyse en fonction d'un rôle potentiellement différent ou moins immédiatement pertinent pour le SCEQE dans la politique de décarbonation industrielle. L'attention particulière portée à l'EU ETS dans cette thèse doit donc être considérée comme ne traitant que d'une partie spécifique du défi consistant à équilibrer les incitations à la décarbonation et l'atténuation des risques de fuite de carbone.

En outre, l'analyse ci-dessus a également suggéré que l'ETS est peut-être un instrument mieux adapté pour gouverner la décarbonation des industries à forte consommation d'énergie à long terme qu'à court terme. Si tel est le cas, cela reformule certaines des questions soulevées dans cette thèse sous un angle différent. Par exemple, la répercussion intégrale des prix du carbone ETS sur les prix des produits industriels à forte consommation d'énergie est-elle la question la plus pertinente à court terme? Ou d'autres approches politiques - telles que les normes réglementaires ou même les marchés publics et le soutien aux subventions pour les matériaux innovants devraient-elles être prioritaires à court terme?

## **Implications pour la recherche future**



Les réflexions ci-dessus sur les changements dans le contexte politique environnant soulèvent également de nouvelles questions qui semblent aujourd'hui plus pertinentes au sujet du commerce, de la compétitivité et de la politique climatique. Par exemple:

- Si, comme on vient de le suggérer, un ensemble plus large d'instruments et d'approches politiques est nécessaire pour engager avec succès la décarbonation de l'industrie à forte intensité énergétique, quelles sont les implications potentielles sur le commerce et la compétitivité de ces alternatives? Comment devraient-ils être abordés du point de vue commercial?
- La discussion ci-dessus sur l'ETS suggérait qu'en plus de la tarification du carbone ou d'autres mesures réglementaires, une vision positive des voies pour que les industries à forte intensité énergétique se « réinventent » avec des modèles économiques rentables semble nécessaire. Si cela est vrai, cela revient à demander une sorte de politique industrielle pour les EITE en Europe. Comment cette politique industrielle devrait-elle être poursuivie dans la pratique et les effets potentiels sur le commerce et la compétitivité (intra et extra-UE) gérés?
- La montée du populisme politique a mis en évidence l'importance potentielle des mesures liées au commerce comme sanction, ou du moins comme une menace, contre la non-coopération dans les accords internationaux sur le climat tels que l'Accord de Paris sur le climat. Comment de telles sanctions pourraient-elles être conçues comme une incitation efficace à la non-coopération?
- Plus généralement, dans un monde où les industries commerciales cherchent à décarboner, mais en utilisant une gamme d'outils et d'approches différentes - comme le suggère la conception flexible de l'Accord de Paris - comment gérer les risques plus larges de frictions commerciales? Le monde politique populiste suggère aujourd'hui que ces frictions pourraient être potentiellement nuisibles à la crédibilité de tels accords. Quels efforts de dialogue et de coopération internationaux pourraient être nécessaires pour créer des règles de base largement acceptables pour les questions liées au commerce découlant des efforts d'atténuation des changements climatiques dans les industries mondialisées?

# Introduction

## 1.1. Introduction

In 2010, I decided to write a PhD thesis on the topic of “Trade, Competitiveness and Climate Policy”. The chapters that follow represent the final results of that undertaking. Before moving on to the substance of the research questions that I examined and the detailed results and conclusions that are described in these chapters, some elements of context are necessary. Thus, this introduction aims to address a couple of important “contextual” issues.

Firstly, it is necessary to outline the policy context surrounding climate change in which this work was being done. The topic is, after all, a policy-oriented question and each of the individual chapters of the thesis were responding to a specific policy context. Secondly, having provided this context, an overview of the key results and insights of each of the three chapters is provided. Thirdly, it is also true that the climate change policy context has changed and evolved during the period that this thesis was being written. The views of the author have also changed and evolved during this time. Some space is thus dedicated to looking back retrospectively at the problematic of trade, competitiveness and climate policy, in order to highlight those areas where I believe that the climate policy debate has moved on. In doing so, I will also try to highlight some questions that are becoming relevant for the research agenda to support future policy in this subject area. Finally, this introduction concludes with a brief reflection on the personal research aims of the author in approaching each of these questions and a note on how this informed the methodology that has been used.

## 1.2. The climate policy context in which this thesis was undertaken

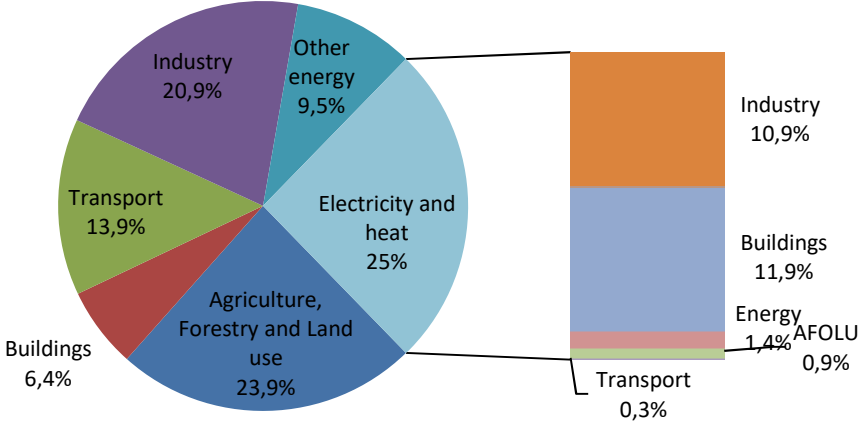
The motivation to write a thesis on the topic of “Climate Policy, Trade and Competitiveness” stemmed from a few simultaneous observations. The first observation was efforts to keep global average temperature rises below 2°C – the threshold commonly used by the Intergovernmental Panel on Climate Change to describe “dangerous” climate change and the objective referred to in all international climate treaties dating back to the Kyoto Protocol – required significant and deep decarbonisation from heavy industry. Industrial production processes account for approximately 21% of direct global Greenhouse gas emissions and 11% of indirect emissions (due to use of electricity and heat production), thus making up 32 % of the global total when these two figures are added (see Figure 1.1).

Most of these industrial emissions come from a small handful of very energy-intensive sub-sectors, with the leading subsectors being cement (13%), ferrous and non-ferrous metals (mainly iron, steel, aluminium and copper (22%)), and chemical production (15%), as well as a handful of smaller sectors such as non-metallic minerals productions (typically things like glass, ceramic, plaster and brick), nitrogenous fertiliser, pulp and paper, wood, rubber, textile production, etc (cf. Figure 1.2).

The Intergovernmental Panel on Climate Change has for a long time suggested that, at least for developed countries as a group, there is a need to get net GHG emissions down to -80 to -95% by around 2050, and decline to zero by the middle of the latter half of the century (IPCC, AR3, 2001 Summary). In light of reduction potentials in other sectors, this effectively means almost full decarbonisation of energy use and production processes for energy intensive industries, like cement, steel and chemicals (cf. IPCC AR4 WGIII Ch10). Therefore the need for an effective, cost-efficient and politically

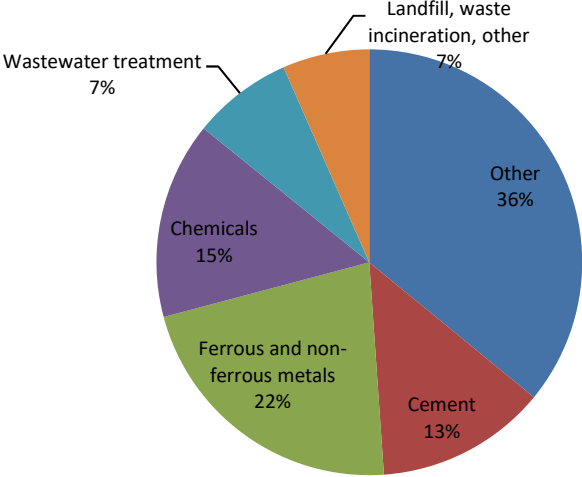
acceptable strategy for tackling emissions from heavy industry was (and remains) a key part of efforts to tackle climate change.

**Figure 1.1 Industrial emissions as a share of total global emissions**



Source: IPCC 2010, [http://ar5-syr.ipcc.ch/topic\\_observedchanges.php](http://ar5-syr.ipcc.ch/topic_observedchanges.php)

**Figure 1.2 Breakdown of global direct GHG emissions by industry**



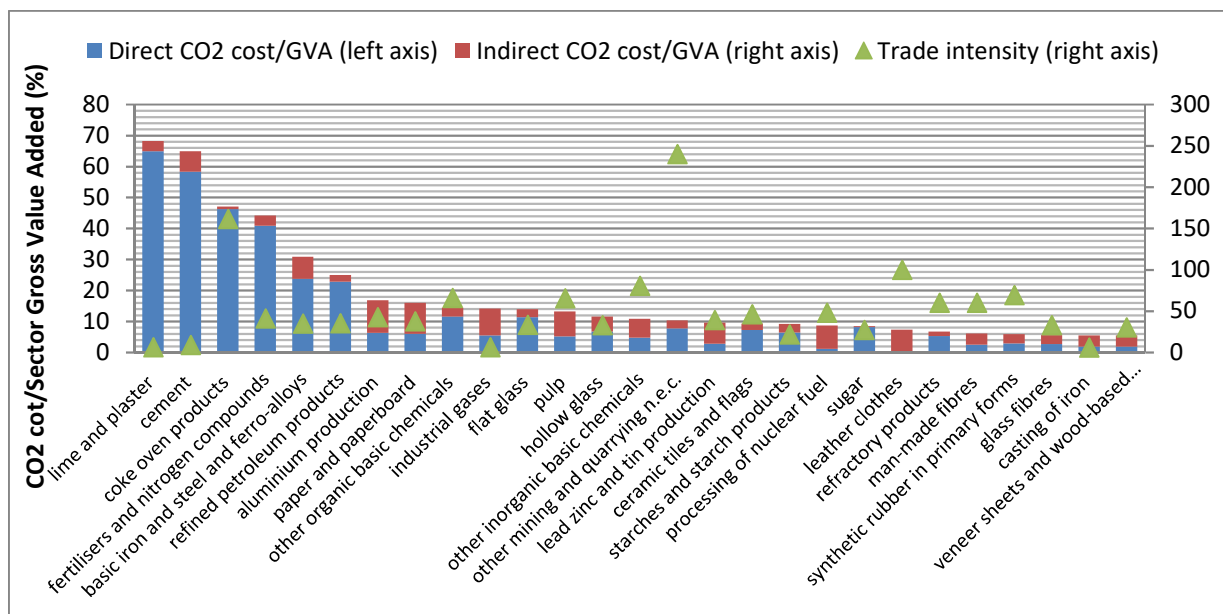
Source: IPCC, 2010, WG3 AR5 Ch 10 [https://www.ipcc.ch/pdf/assessment-report/ar5/wg3/ipcc\\_wg3\\_ar5\\_chapter10.pdf](https://www.ipcc.ch/pdf/assessment-report/ar5/wg3/ipcc_wg3_ar5_chapter10.pdf)

A second and related observation was that energy-intensive industry tends to produce products that are, or can be, traded in international markets. The combination of high emissions intensity plus tradability across national borders means that the incremental production cost induced by regulation aiming at decarbonising production could inadvertently lead to perverse outcomes - notably delocalisation of production, investment, employment and emissions to less regulated parts of the world. Although

sometimes exaggerated by industry when opposing regulation, or by climate change denialists when arguing that action is too costly, for a small handful of energy intensive sectors this risk – known in the jargon as “carbon leakage” – is real.

The nature of this risk can be seen in Figure 1.3. The figure uses EU industry and trade data. It then assumes a CO<sub>2</sub> price of 30€/tonne and calculates what the additional cost to specific energy intensive industries would be as a share of their gross value added margin. The blue bars represent the direct emissions cost, while the red bars represent the indirect emissions cost (from electricity consumption). This gives an indication of the cost-sensitivity of industries to additional CO<sub>2</sub> regulation. Meanwhile, the green triangle series shows for each industry a measure of “trade exposure”. This indicator is measured as the ratio of imports plus exports to domestic production plus imports (between the EU and non-EU countries in this case). It gives a sense of how easy it is for the products in the industry to be traded internationally and thus an indicative (if imperfect) measure of the potential for loss of competitiveness to delocalised production. The results show that there are a handful of around 12-15 sectors with relatively high CO<sub>2</sub> costs as a share of value added at 30€/tonne. Meanwhile, most of these sectors have a non-negligible level of international trade. One exception to this is cement and lime and plaster production. However, it should be noted that in this case the lack of tradability reflects high weight to value added ratios, which induce high transport costs. Analysis has however been done which suggests that at sufficiently high CO<sub>2</sub> prices (roughly 25€ or above) transportation by ship from abroad to coastal regions can become economical. Thus, the risk of carbon leakage at meaningful carbon prices of 30€/tCO<sub>2</sub> or above – i.e. price levels which are often thought modest compared to prices needed to drive significant decarbonisation – can become significant.

**Figure 1.3 Who is an energy intensive trade exposed industry?**



Moreover, countries around the world will most likely continue to pursue domestic decarbonisation objectives at different speeds and in different ways. Thus, the potential for significant and sustained marginal production cost differentials that impact trade flows for so-called “Energy-intensive Trade-exposed industries (EITEs) between nations is also real.

It was therefore clear that efforts to decarbonise energy intensive trade exposed industries would need to find innovative and economically intelligent solutions to the problem of decarbonising energy intensive industries in the face of the threat of carbon leakage. These solutions needed to be effective at eliminating leakage risk while also maintaining incentives to decarbonise, be as economically efficient as possible, and broadly politically acceptable both to industry and civil society groups that opposed excessive generosity to industry. This global dimension makes energy intensive industry different from a policy perspective in an important way from many other major emitting sectors, such as power production, heating and cooling in buildings, and road transport, where the barriers to trade across national borders are typically prohibitive.

A fourth and final key observation was that in the late 2000s and early 2010s so-called “emissions trading” instruments were also seen by most experts and environmental economists as the main policy vehicle for driving this decarbonisation of the energy intensive and other key industries, such as power production. For instance, the OECD, which generally reflects the conventional policy wisdom among developed countries, stated that “a broader use of emission trading systems (or of environmental taxation) would be one of the most efficient and effective ways of promoting green growth” (OECD.org, 2013)<sup>1</sup>. In this context, the EU developed and began the EU Emissions Trading Scheme (EU ETS) in 2005, and many other jurisdictions at both national and sub-national level – such as Australia, New Zealand, Japan, South Korea, China, and several US states including California and New York, were developing their own ETS in the late 2000s and early 2010s. In this context the EU ETS was thus considered by many to be not only a main pillar of efforts to tackle climate change in the EU, but also considered a vital testing ground for the future of climate policy at the global level.

However, from early on in the EU ETS and when this thesis was being conceived, it was evident that, to succeed, the EU ETS – and other emissions trading schemes like it – would need to find a satisfactory and lasting solution to the question of carbon leakage risk. Otherwise, these instruments would tend to die a slow death through their inability to garner sufficiently broad support amongst key stakeholder groups.

Indeed, concerns about adverse impacts on international competitiveness and carbon leakage were one of the main arguments put forward – whether in good faith or bad – to justify resistance to more ambitious climate policy. A good example of this is the EU ETS itself. Even after 5 to 7 years of experience with the EU ETS, creating a functioning emissions trading system was proving challenging. This difficulty has been partly due to opposition from member states with coal intensive power sectors, such as Poland, where approximately 90% of electricity is still generated from hard coal and lignite. However, at least as much opposition has also come from energy intensive trade exposed industry. To cite just one example among many, in 2005, when the ETS was launched, the European cement association, Cembureau, argued that “the unilateral initiative of the EU to launch its ETS is leading to major problems of competitiveness of the EU cement industry and will not succeed in reducing global CO<sub>2</sub> emissions” (Skjaereth and Eikeland, 2016)<sup>2</sup>. Since then the cement sector has repeatedly opposed efforts to strengthen the incentives created by EU ETS to decarbonise its sector, often the basis of competitiveness concerns. This included opposition to the repeated efforts to address important problems that have arisen in the context of the EU ETS, such as, for instance, a very large structural surplus of emissions allowance, which have weighed on prices since the global economic and eurozone debt crises of 2009 and 2012 (Influence Map, 2017). Other EITE sectors have typically adopted similar positions. Thus, the problem of how to find the right policy solutions that would make the carbon leakage

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<sup>1</sup> <http://www.oecd.org/env/tools-evaluation/emissiontradingsystems.htm> (last accessed: 28/07/2017)

<sup>2</sup> Corporate Responses to EU Emissions Trading: Resistance, Innovation Or Responsibility, Pg 173

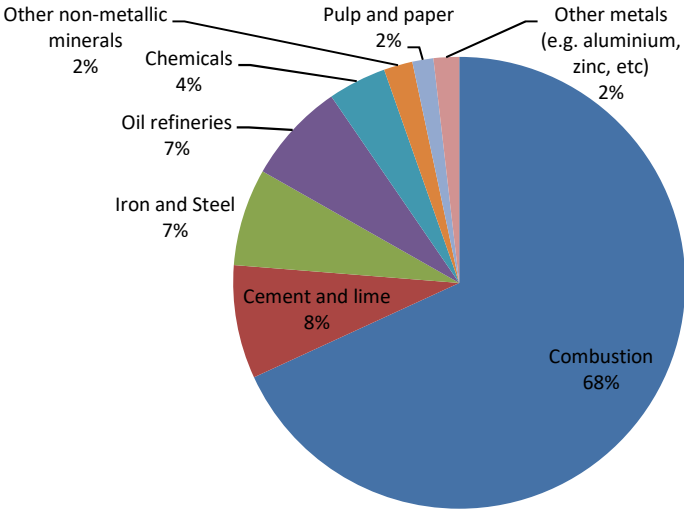
argument against ambition “disappear” within the context of emissions trading systems appeared to be of paramount importance as a policy research question when this thesis was conceived.

**Box 1.1 The EU ETS and its relevance to EU and global climate policy**

The EU Emissions Trading Scheme is a multi-national system for limiting and phasing down Greenhouse Gases in Europe. It is the first large scale “cap and trade” scheme for greenhouse gases to be put in place anywhere in the world. As such it has been the subject of intense interest by climate policy researchers. It operates in the EU28 member states and including 3 additional members of the European Economic Area (Norway, Iceland and Liechtenstein). Installations with a thermal rated capacity of 25MW or more are typically obliged to participate in the scheme. The main activities covered by the scheme are combustion for power production and district heating networks, energy intensive industrial production processes, such as cement clinker production, ferrous and non-ferrous metals productions, oil refining and production of certain chemicals, as well as other activities such as glass, pulp and paper, brick and ceramics production. From 2012, emissions attributed to intra-EEA flights from airlines have also been included in the scheme. The system now covers approximately 12,000 point source installations within the EU.

The scheme is organised in “compliance phases”: Phase 1 or Trial Phase (2005-2007), Phase 2 (2008-2012), Phase 3 (2013-2020), Phase 4 (2021-2028). As a world first pilot scheme, each Phase has been accompanied by lessons learned and a subsequent round of reforms to try to improve the functioning of the Scheme in subsequent phases.

**Figure 1.4 Sectoral composition of ETS installations by main activity**



Source: EEA, 2016 Trends and projections in EU ETS

Note: “Combustion” category refers mainly to electricity and heat producing installations for residential and the tertiary sector. However it also includes combustion activities related to the production of other categories in the above graphic, such as cement and lime, iron and steel, and chemicals. Thus, the figures above tend to understate the size of these smaller categories.

The system operates by placing an absolute limit or « cap » on the aggregate combined emissions of the covered installations and issuing emissions permits or “European Union Allowances” equivalent to that

limit each year. Following each calendar year firms with covered installations are legally obliged to submit an equivalent number of allowances matching their verified emissions in the previous year. Failure to do submit the correct number of allowances can lead to a fine of 100€ per missing allowance and a requirement to make good on the shortfall in the following year. The economic principle underlying the scheme is that by progressively issuing a smaller number of allowances to covered entities each year, a scarcity of emissions rights is created. Firms are freely allowed to trade allowances between each other depending on their compliance needs. The combination of scarcity and tradability of allowances enables an allowance or “carbon” price to emerge in a “carbon market”. This price in turn allows firms to optimise their abatement decisions in the short term, by choosing to buy allowances or abate and sell allowances depending on their private marginal abatement costs.

Figure 1.5 below shows the EUA carbon price between 2005 when the scheme began and early 2017. It can be seen that the price has fluctuated significantly. These price fluctuations have been due to a few factors. The large fall and recovery in the price during the Phase I of the Scheme was due to the Trial nature of the scheme at that time. Due to data uncertainty about actual historical emissions prior to the creation of the Scheme, the EU was forced to rely on unreliable estimates of emissions from several EU countries. This led to an oversupply of allowances during the Trial phase and, as allowances could not be banked into future Phases, the price equilibrium after some abatement occurred was zero. In Phase II of the scheme the price began again at around 28 €/tonne CO<sub>2</sub>, only to fall once following the collapse in industrial and power production following the macro-economic crises of that period. This was also exacerbated by provisions in the design of the scheme which allowed for a large number of international offset credits to be further added to the existing oversupply supply of allowances. Continued improvements in energy efficiency, rising shares of renewable energy, and weak industrial production and closure of overcapacity in several energy intensive industrial sectors in the EU since then have all contributed to a large and sustained oversupply of allowances since then, leading to weak allowances prices and very limited incentives for abatement.

**Figure 1.5 EU ETS CO<sub>2</sub> allowance price evolution (Jan 2005 to July 2017)**



Source: Author, based on data from ICE and Investing.com

**1.3.Key insights from the specific research questions addressed**

The research contained in the chapters that follow revealed a number of interesting insights. These insights should be considered part of the broader policy literature, and especially ex-post policy analysis literature, on Emissions Trading Systems. Thus the focus is highly policy oriented and designed to be

consumable, to the extent possible for a PhD thesis, by policy experts and policy-makers. As a consequence, the methodological approach has aimed to strike a balance between intellectual and economic rigour from and readability less expert people in the climate policy field.

Each of the three chapters that follow addresses a separate question and was initially written for independent publication. However, the three chapters are connected by a common thematic objective. Specifically, each of these questions aims to respond to the problem of how to combine effective incentives for decarbonisation of energy intensive trade exposed industry while simultaneously minimising the risk of carbon leakage.

Chapter 2 looks specifically at the introduction of a new reform for carbon leakage exposed sectors in the EU ETS in 2012/2013, in the form of free allowances based on emissions performance “benchmarks”. The issue of benchmarks was immediately interesting for my topic because it was an attempt by the EU to respond to a set of criticisms and complaints by different stakeholder groups during the ETS trial phase. These criticisms related to the way that energy intensive industry and carbon leakage risks were being handled by the design of the ETS itself. Concerns included the fact that the flexible approach to letting member states choose their free allocation systems was leading to massive welfare transfers to certain installations or industries; that the internal market was potentially being distorted; and that incentives for abatement were being diluted.

To address these concerns the EU therefore responded by implementing a system of free allocations to carbon leakage exposed sectors based on pre-determined best available technology benchmarks. The premise was that only the top 10% of installations in a given sector would receive 100% of their allowances for free, thus creating a target for laggard installations to aim at. At the time, this was the first time such benchmarking had been used for industry in climate policy.

This chapter was written precisely at the time when the first data on the new mechanism was being released (in 2012). My co-authors and I therefore sought to make use of these new detailed and relatively high quality new data on individual ETS installations to analyse what the new system meant. Specifically, we sought to be the first to use these data to compare allocation rules in Phases 1 & 2 vs. Phase 3 and their implications. Bearing in mind the cost and complexity of developing and administering the new mechanism (~25-30 million €), 3 criteria were evaluated:

- the welfare distributional implications of the change in the rules;
- the adequacy of protection from carbon leakage risk, and
- the risk of internal market distortions that was eliminated or that remained due to the system.

We found that the new data coming out and which we carefully collated and analysed did indeed allow for a more precise evaluation of the impacts of free allocation rules in EU ETS Phases 1 and 2 and, although just starting, some interesting predictions and insights could be drawn about Phase 3.

Specifically, we were about to conclude that the move to harmonised benchmarking had indeed brought significant benefits over grandfathering system. Most notably it had led to a significant reduction of unjustified subsidies to European industry worth several tens of billions of Euros during Phase 3 alone under expected carbon prices. The word unjustified here refers to the fact that these free allocations were in excess of total emissions and therefore were effectively “overcompensating” for the newly incurred costs of regulation. In addition to this benefit, the new rules also introduced more transparency in the methodology behind the free allocations, and coupled with the use of a (more or less) common rulebook



for all installations in Europe, this in effect created a system that both was but was also broadly seen as being fairer by stakeholders (give or take some remaining weaknesses, cf. Chapter 3).

However, the study also highlighted some weaknesses of the new mechanism. Firstly, we found that it may protect poorly against leakage at higher CO<sub>2</sub> prices (e.g. at CO<sub>2</sub> prices above 25 €/tCO<sub>2</sub>) or indeed in a growth context in which historical production levels (on which total allocations are based) are surpassed by actual production. The mechanism also was limited as a driver of low-carbon technology innovations, since it did not provide for any means of paying for the incremental costs of innovation about the carbon price. With the benefit of hindsight, this is and remains a major weakness of the incentive structure created for industrial decarbonisation under the ETS + benchmark-based free allocation approach.

To be precise, there were also some weaknesses due to specific design aspects of the way the EU implemented the benchmarking approach and just not benchmarking per se. For example, the EU chose that allocations would be based on a benchmark multiplied by “ex-ante” production levels, coupled with a set of thresholds triggering “clawbacks” from future free allocation levels (an important design floor of the approach discussed in detail in Chapter 3). Furthermore, the EU further complicated the politics and economics of its benchmark-based allocation system by making the choice to apply a so-called ‘cross-sectoral correction factor’ to the free allocations to effectively keep the total allocation pie from growing as a share of the system cap on emissions. Both these choices effectively meant that future allocations, even with benchmarks, could and would still deviate, sometimes considerably, from actual emissions. In hindsight, these decisions have proved highly controversial. They have also, in this author’s opinion, been detrimental to the policy objective of negating the “carbon leakage argument” as a barrier to strengthening the carbon price within the system itself and thus driving faster and deeper decarbonisation.

Finally, the paper provided some other interesting insights about how EU member states effectively went about giving out free allowances under the more flexible rules prior to the benchmarking system’s introduction. Effectively member states were not just grandfathering allowances, as some in the literature (e.g. Ellerman et al, 2010) have suggested. Rather, “grandfathering” was not just grandfathering historical emissions levels, it was grandfathering historical emissions plus an extra chunk of allocation. This can perhaps be interpreted as an insight into how countries in a common emissions trading scheme can be incentivised to act, when the desire for supporting national industries coincides with the creation of a new source of value (emissions certificates).

While this situation created a wasteful free-for-all for industry, we nevertheless found that it was hard to justify the cost of implementation of the new system purely in terms that it created a more level playing field for companies within Europe. While this was true, since many member states tended to over-allocate their industries by similar percentage amounts, and since trade costs of many of the goods concerned (e.g. cement clinker) were quite high relative to implied carbon costs per unit of product, the overall impact on the level playing field of the EU’s internal market was not a major problem that needed to be addressed – or at least, not at this phase of the scheme.

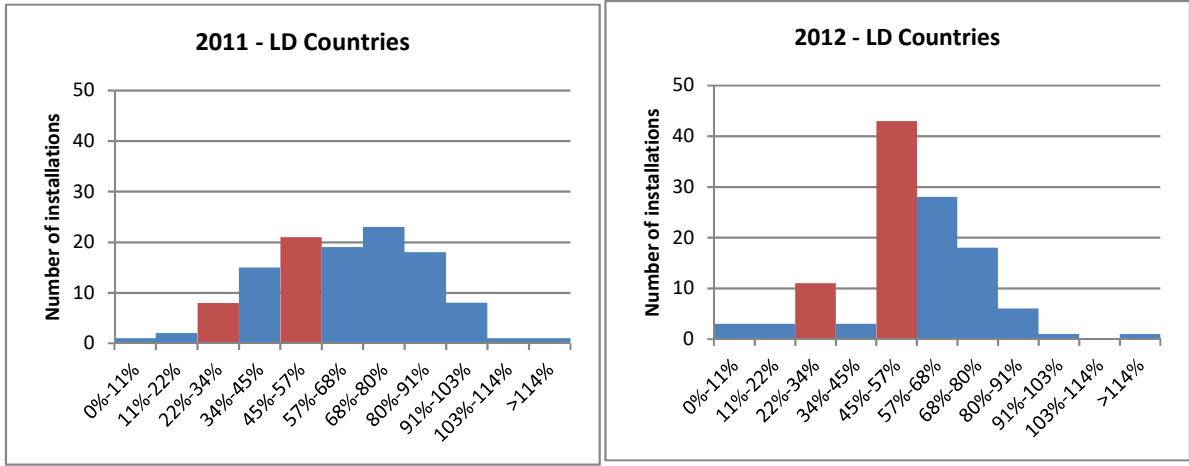
Chapter 3 of the thesis looks at a further important change in the EU-ETS phase 3 for EITs, which concerned the introduction of the “activity level threshold” rule (ALTs). The underlying rationale for its introduction is that it would reduce the over-allocation profits to industry receiving free allocation in case of downfall in the 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%). However, the concern of my co-authors and I was that by creating such thresholds, strong incentives would be created to game the system. We therefore decided to bring this issue to the data and provide the first evaluation of the rule in the literature.

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). The cement sector was a good candidate sector to look at empirically, since it has a simple technological link between emissions and production levels that made the use of proxy variables to estimate production more reliable. It was also a sector with very high carbon intensity per unit of production, making the economic incentives to game the system potentially very strong under certain circumstances. “Cement in 2012” thus provided a natural experiment to evaluate the consequences of this rule.

Our main conclusion is that while ALTs did reduce to some extent over-allocation profits, it also created significant operational distortions which led to outcomes inconsistent with the low carbon transition of EU energy intensive industries. Specifically, it showed that there was strong evidence of threshold effects, whereby cement firms find ways of increasing clinker production (the most emissions intensive part of the cement process) above what they would otherwise have produced in order to get across the threshold and gain an extra 50 to 25% of allowances in their next year’s allocation. We identified 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.

**Figure 1.6 Number of cement installations producing at different levels of capacity in 2011 (no threshold rules) vs. 2012 (thresholds rules enacted)**



Note: The Figure shows that the enactment of the thresholds rules leads to a significant alteration in the distribution of firms’ production levels, with more firms producing just above the 25% and especially the 50% threshold level (see Chapter 3 for more details). Note that this result is all the more striking since aggregate demand in the countries in which these installations were operating fell between 2011 and 2012...

An obvious consequence is also that the reduction in over-allocation profits is also less than expected because of the gaming behaviour of the industry to achieve the thresholds, during periods of low market demand.

The operational distortions reported in our study 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. 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.

The evidence that the threshold effects were causing significant distortions was very strong even without detailed counterfactual analysis (see Figure 1.6 for instance). A more difficult question is: what was the size of the distortion? And: by how much did it undermine the Commission's intended objective of limited excessive allocations to firms that had significantly reduced operations? To answer this question, a counterfactual scenario was developed.

With the elaboration of a counterfactual, we estimated that in our subset of "low demand" countries in the EU, clinker production had increased by around 7.2 Megatons in 2012, and that emissions in these countries had therefore increased by around 6.4 Megatons of CO<sub>2</sub>. Evidence of such increases in higher demand countries were somewhat weaker however, suggesting such behaviour was marginal at best (as would be expected). To put this in perspective, this compares to a clinker market of ~134 Mt in 2012 and total cement sector and EU ETS emissions in 2012 of approximately 113 MtCO<sub>2</sub> and 1800 MtCO<sub>2</sub>eq respectively. The distortions, if correctly estimated, are therefore meaningfully large for the cement sector (~+6% of emissions/production) and quantitatively large in absolute terms, although they are modest compared to the size of the emissions currently covered by the EU ETS (which, after all, covers ~45% of total EU CO<sub>2</sub> emissions).

In terms quantified that after the introduction of ALTs: the potential over-allocation profit for the EU's cement sector 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 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. This would suggest that the Commission's effort to curb free allocation windfalls to the cement industry at least using this approach has been only marginally effective, while also inducing arguably quite negative environmental, economic (and potentially political) side effects.

Finally, it's worth noting that 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.<sup>3</sup> To be sure, had we observed demand growth, the threshold rule may have been less relevant. However, the threshold effects may then have been on the other side of the equation, with the reserve for new entrants may have been a more important source of distortions (there would be an incentive to have a artificially high production during the period used to fix the equivalent of HAL for new entrants).

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<sup>3</sup> Take a EUA price at 20€/t a simple extrapolation for LD countries would bring up the potential wind fall profit to  $236 \cdot 20/9 = 524$  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.

Now, the construction of the counter-factual scenario is obviously dependent on the available data. Ideally one would have liked to have had for each ETS installation in the sample of low demand countries an identical cement installation just next to it that was not included in the EU ETS for which production and emissions data was available. Unfortunately, such a “matching analysis” was not feasible due to a lack of non-ETS cement plant in the relevant countries and a lack of data on their production for those very few that exist. For the installations we had, we ran a large number of variables for which data were available at the plant level (including location characteristics, differences from the country mean in previous years, previous year’s activity level, company dummies, etc) in order to get the best possible fit for past observed data. In the end, we were able to arrive at a simple model that explained a very respectable 85% of the variation in installations production levels. Although our estimates may not be extremely precise, we were nonetheless confident that the orders of magnitude are broadly an accurate reflection of what went on in 2012.

This confidence emerged in part from the confidence we have in the counterfactual model itself, together with the fact that several different pieces of statistical analysis we provide (trade data, company-plant level data, and the raw statistical analysis of the numbers) all seem to point to effects of the magnitude we find in the counterfactual results. However, in addition to the quantitative evidence presented here, we later interviewed cement sector executives and operational managers who confirmed that this “gaming” of the threshold rules<sup>4</sup>. Their own comments on our results did not seem to catch much doubt on either the existence of the gaming behaviour or the rough orders of magnitude we found.

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 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

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<sup>4</sup> Cement sectors executives unfortunately asked not to be named in our study, thus they are not referred to in the bibliography to the study.

analysis in this paper suggests that this option would outperform both ex ante allocation with and without thresholds in terms of reducing distortions and over allocation profits.

However, we also argued that 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 on-going 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 2014a). Another alternative may be to forget about free allocation and implement a border tax adjustment; however this also comes up against major political and diplomatic challenges.

Chapter 4 of the thesis looks at the question of whether ETS firms in so-called energy intensive trade-exposed industries pass on the carbon cost of their production process to consumers in the price of their products. The chapter begins by noting that recent studies of emissions reduction potentials for some large EITE sectors like cement and steel have demonstrated that, based on current technology, it is impossible to achieve sufficiently deep cuts in emissions without substantial improvements to the way society not only produces, but also consumes, carbon intensive materials (Neuhoff et al, 2015a; Neuhoff et al, 2015b). In short, we will need:

- much greater efficiency in use of these materials by consumers,
- greater substitution between high and low-carbon materials in appropriate applications, and
- stronger economic incentives for product innovation that reduces the carbon footprint of the materials used.

In principle this is a role that the EU ETS should play. After all, there is also substantial complexity of the range of products, of end-use applications and of potential substitutes along the value chain in carbon intensive material sectors. This complexity makes it seem unlikely that appropriate levels of product substitution, material efficiency, and innovation support, could be mandated by a centralized regulator in an efficient and politically durable way. In short, a robust carbon price that is passed along the value chain in energy-intensive materials sectors seems necessary.

This in turn raises the question of whether carbon prices are indeed passed on in product prices by EITE industries. This chapter notes that the evidence on this question is far from complete. It also notes that basic economic theory of opportunity costs suggests that price pass-through is a matter of course and that the European Commission tends to use this theoretical argument to justify attempts to cut back on free allocation to industry to protect against carbon leakage. On the other hand, it notes that there are, in the policy debate, plausible sounding arguments that are sometimes put forth about why the textbook theory of price pass-through even in the presence of freely allocated emissions allowances might not be true in practice. The question therefore once again had strong relevance for anti-leakage measure design under the EU ETS.

I therefore sought to answer this question by looking at two international traded sectors in the EU ETS which have among the highest CO<sub>2</sub> costs per unit of production, namely cement and commodity steel.

The implicit assumption is that if EITE sectors are passing on their CO<sub>2</sub> costs to consumers then these two sectors are so intensive in CO<sub>2</sub> per unit of product that the evidence should be quite obvious.

After a great deal of econometrics and statistical analysis, the result I found was troubling for defenders of the current system of anti-leakage protection under the EU ETS. In short, the chapter showed that in fact there has been no pass-through of EU ETS carbon prices into cement and steel product prices, or if there has been it has been so margin as to be practically useless as a source of incentives for consumers to reduce their material demand, or substitute, or be willing to pay for more expensive innovative versions of the same products.

The implications of this finding are very strong. Essentially, it implies that the EU ETS, even if it were to overcome its current problems in terms of allowance oversupply and lack of bankability for investors, would nevertheless still be ineffective at driving the decarbonisation of energy intensive industry as currently designed. To overcome this weakness, it would need to make significant reforms also to entire approach to avoiding carbon leakage. In short, ex-ante free allocation would need to be abandoned and either system of border adjustments and full auctioning or ex-post free allocation together with a downstream carbon charge would be needed. This is disconcerting as it implies a very radical break with the current thinking in large parts of the climate policy world.

## **1.4. An evolving policy context**

Since the period when this work was begun, the climate policy debate and the surrounding policy context has also changed considerably and in some ways that were unforeseen by the author when this work was begun. For one thing, more experience has been gained with climate policy and with Emissions Trading as an instrument of climate policy in Europe and elsewhere. Further, the overarching international climate policy context has taken some important intellectual leaps forwards (such as with the Paris Agreement) and backwards (such as with the rise of political populism in the United States and Europe). The rise of political populism has made international trade a much more politicised and polarising issue, with important ramifications for climate policy. Furthermore, in the wake of a more divided European Union, it has become increasingly evident that the EU institutions will struggle to govern pan-European climate and energy policy as easily as once might have been assumed by the creators of the EU ETS. In the climate and energy sphere, more emphasis is now being placed in expert discourse on supporting regional sub-groupings of “leaders” and EU “enabling” of national or regional strategies to tackle thorny policy problems.

These developments are all relevant to the topics addressed and conclusions drawn in the individual chapters of this thesis because they throw into question some of the implicit assumptions that underlie the work. For example, for the reasons highlighted above, this thesis places a strong focus on carbon pricing and emissions trading in particular as a vital and potentially even the main pillar of efforts to decarbonise energy intensive industry. Today, with the benefit of hindsight, this assumption deserves to be questioned much more seriously.

One important weakness of the EU ETS that has been revealed with the benefit of time is that it is unlikely to be able to deliver sufficient incentives for investments in capital intensive low carbon technologies. There are a few reasons for this. One is that the price evolution of the EU ETS has shown that the market is, in effect, incapable of delivering a relevant price signal that reflects the long term carbon scarcity. Experience has revealed that prices are almost entirely a reflection of short term balances between supply and demand in the market, while long term price fundamentals have little to

no impact (Schopp et al, 2012; Sartor et al, 2013). Since many deep decarbonisation technology investments for energy intensive industry have financial lifetimes of 20 years or more, the EU ETS is clearly not sending the necessary price signals to industry to decarbonise in a timely manner.

Another weakness of the EU ETS (and of ETS instruments more generally) is that financiers typically require other policy guarantees or justifications for the business case underlying any given low carbon investment *in addition to any EU ETS carbon price*. This appears to be due to limited historical price experience to base modelling on, high perceived political risk which is difficult to price, and now, unfortunately, a poor track record of generating meaningful carbon prices. Thus most financing companies for large investments in low carbon technology still struggle to assign to integrate CO2 prices from the EU ETS into their lending models. Consequently, *even if* long term carbon scarcity signals could be delivered by the market to incentivise industry to invest in capital intensive decarbonisation technologies, they would still have trouble financing the deployment and commercialisation of their investments.

Furthermore, in the case of Europe, the EU ETS experience has demonstrated the extreme difficulty of generating and maintaining the necessary political support to even deliver a scarcity of allowances to an ETS in the short run. This has been amply demonstrated in the supreme political difficulty that was encountered in the EU in order to address a massive structural surplus of emissions allowances that was generated during the Great Recession. Thus, in practice, carbon prices have remained more or less insignificant to provide any form of incentive for abatement, let alone large scale investments in relatively expensive investments in new low carbon capital equipment (like CCS) or investments in commercialisation of innovative substitute materials.

This political difficulty of generating meaningful economic incentives for decarbonisation in an ETS is partly a European problem – given the challenge of agreeing between 27 or 28 different member states with very different economic and emissions profiles. However, the EU is not alone in this. Other countries such as Japan, progressive states of the United States and Canada, and Australia and New Zealand – all once hailed as likely to follow the EU into the emissions trading club – have since seen early progress towards ETS markets slow or reverse. This phenomenon has various causes. However, in this author’s opinion a common factor across many of these cases is that carbon pricing and emissions trading schemes in particular ignore fundamental political economy concerns of key stakeholders. Most notably, they are typically implemented with a focus on their “technological neutrality” and “economic optimality for society as a whole”, but policy makers then give little attention to the revolutionary structural changes that are being required of industries to completely change their technology, capital stock and business models. Thus, ETSs are often introduced without an accompanying narrative or vision of how specific energy intensive industries can transform themselves to survive and be profitable in a decarbonised world. Not surprisingly, they are thus perceived by industry as punitive measures that are a direct attack on their ability to survive and perform financially.

Moreover, this occurs in a context where globalisation and the Great Recession have already placed enormous pressure on the finances of energy-intensive commodity industries in many parts of the world. In short, even as a self-professed “card-carrying economist”, this author is increasingly convinced that climate action to decarbonise energy intensive industry has actually been hurt a little bit by a headlong search for economic purity – in the form of emissions trading markets – and a failure to keep an eye on important social and political forces that shape the surrounding political economy of bringing about lasting change.

Now, this does not mean that the author believes the EU ETS to be irrelevant for decarbonisation of industry. The EU ETS still has a few potentially important advantages for attacking industrial

decarbonisation that cannot be ignored either. One advantage is that it provides an absolute cap on GHG emissions for a broad set of sectors of the economy, including energy intensive industry. In the long run, this is likely to become increasingly important, as the total emissions budget tightens and the issue of a level playing field across countries and industries participating in the EU's single market will become more relevant. Another advantage is that the EU ETS is, at least in principle, its ability to provide coherent incentives throughout the value chain of energy intensive production. There are, after all, many vectors for decarbonising industry, from production process technology and fuel switching, on the supply side, to increasing end use efficiency and substitution on the demand side. As noted in Chapter 4 of this thesis, it remains difficult to see how all of these vectors, especially those on the demand side, could be regulated in a simple, thorough and non-distortionary way without the aid of pricing instruments. Of course, in the short run, one could imagine sacrificing some efficiency and living with certain distortions for the sake of effectiveness. In the long run, however, such approaches may also come up against their political economy limitations.

So, what does this analysis mean for how one should interpret the conclusions in each of the following chapters? Firstly, it means that one needs to interpret the parts of the analysis with respect to a potentially different or less immediately relevant role for the EU ETS in industrial decarbonisation policy. The intensive focus on the EU ETS in this thesis therefore needs to be seen as addressing only one specific part of the challenge of balancing decarbonisation incentives and carbon leakage risk mitigation.

Furthermore, the above analysis also suggested that perhaps the ETS is better adapted instrument to governing the decarbonisation of energy-intensive industries in the long run than in the short run. If this is the case, then this recasts some of the questions raised in this thesis in a different light. For instance, is full price pass-through of ETS carbon prices into energy-intensive industrial product prices the most relevant issue in the short term? Or should other policy approaches – such as regulatory standards or even public procurement and subsidy support for innovative materials take priority in the short term?

## **1.5. Implications for future research**

The above reflections on the changes in the surrounding policy context also raise new questions that today appear more pertinent to the topic of trade, competitiveness and climate policy. For instance:

- If, as just suggested, a wider suite of policy instruments and approaches are needed to successfully engage the decarbonisation of energy-intensive industry, then what are the potential trade and competitiveness implications of these alternatives? How should they be tackled from a trade perspective?
- The above discussion on the ETS suggested that in addition to carbon pricing or other regulatory measures, a positive vision of pathways for energy intensive industries to “re-invent” themselves with profitable business models seems necessary. If this is true, then this amounts to a call for a kind of industrial policy for EITs in Europe. How should such industrial policy be pursued in practice and the potential trade and competitiveness impacts (both intra and extra-EU) managed?
- The rise of political populism has highlighted the potential importance of trade-related measures as a sanction, or at least a threat, against non-cooperation in international climate agreements such as the Paris Climate Accord. How might such sanctions be designed to work as an effective dis-incentive to non-cooperation?
- More generally, in a world in which traded industries are seeking to decarbonise, but using a range of different tools and approaches – as implied by the flexible design of the Paris



Agreement – how should broader risks of trade frictions be managed? The populist political world today suggests that these frictions could be potentially harmful to the credibility of such accords. What international dialogue and cooperation efforts might be required to create broadly acceptable ground-rules for trade-related issues emerging from climate mitigation efforts in globalised industries?

## Chapter 2

### **Free allocations in EU ETS Phase 3: The impact of emissions-performance benchmarking for carbon-intensive industry<sup>5</sup>**

#### **2.1. Chapter Introduction**

##### **2.1.1. Policy Context**

One of the most controversial aspects of the European Union Emissions Trading Scheme (EU ETS) during its first eight years has been the question of whether or not, and if so then how, to initially allocate emissions allowances to covered installations. Despite the early theoretical result of Montgomery (1972), who argued that initial allocation decision should matter for welfare but not for the economic efficiency of an ETS, a substantial literature has emerged since the creation of the EU ETS debating the relative merits of alternative initial allocation approaches.

This paper contributes to this literature by presenting a first detailed empirical analysis and evaluation of some of the key welfare and efficiency properties of the new free allocation rules for non-electricity sectors in the EU ETS in Phase 3(2013-2020). It exploits an original database compiled by the authors to focus on the question of harmonisation of allocations across the EU, which was one of the key critiques of allocation methods in Phase 1 (2005-2007) and 2 (2008-2012). It asks whether or not the introduction of harmonised benchmarks adequately addresses the potential for distortions to both the carbon market and internal product markets which has been highlighted in earlier literature. It is shown, via the example of the cement sector, that despite meaningful improvements in Phase 3, considerable scope remains for internal market distortions. On the other hand, benchmark-based allocations are found to be a significant improvement on Phase 2 National Allocation Plans (NAPs) in terms of their distributional welfare consequences in several ways.

##### **2.1.2. Welfare, efficiency and the optimal initial allocation literature**

The literature on the initial allocation of pollution licences has generally viewed free allocation as (at best) a second-best approach to initial allocation in the presence of carbon leakage risks. This view has been based on both distributional welfare and economic efficiency arguments. The idea that initial allocations can have important distributional implications goes back to Montgomery (1972), who demonstrated that free allocations have welfare consequences because of the distributional effects of the initial licence endowments, but, in the absence of transaction costs, should not change the efficient market outcome. Since Montgomery, understanding of other kinds of welfare implications of

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<sup>5</sup> This chapter was co-authored with S. Lecourt, then at the Chaire Economie du Climat and Université Paris Dauphine & C. Pallière, then working for CDC Climat Research. It has since been published in *Climate Policy* as: O. Sartor, C. Pallière & S. Lecourt, *Climate Policy* (2014): Benchmark-based allocations in EU ETS Phase 3: an early assessment, *Climate Policy*, DOI: [10.1080/14693062.2014.872888](https://doi.org/10.1080/14693062.2014.872888).

grandfathering allowances has improved. Bovenberg et al (2002), Sijm et al (2006), and Smale et al (2006) have shown that, where companies pass on the opportunity costs of pollution licences into consumer prices, 100% free allocation leads to windfall profits for polluting industry. Furthermore, Crampton and Kerr (2002) and Hepburn et al (2006), Ahman et al (2006), Palmer et al (2006), Kruger et al (2006), Benz et al (2009) have argued that a higher share of initial auctioning is better for aggregate welfare, all else equal, since it pre-empts rent-seeking lobbying costs over the initial division of allowances.

In the current climate change policy context, where carbon prices are unequal in different parts of the world for sectors competing in global markets, free allocation to carbon-intensive sectors is sometimes considered a second-best option to mitigate carbon leakage (Hepburn et al, 2006; Buchner et al, 2007; Ellerman et al, 2010). Against this, a number of authors have persuasively argued that a combination of full auctioning and a border carbon adjustment (BCA) is, in principle, the most efficient and environmentally effective way to mitigate leakage, e.g. Monjon and Quirion (2010). But since BCAs are often thought to be diplomatically challenging to implement, much of the literature, still focuses on how to improve free allocation methods to make them more equitable and efficient.

In this context, the highly criticised decentralised National Allocation Plan (NAP) approach in Phases 1 and 2 has underlined the importance of ensuring that such “second-best” free allocation approaches be applied well. The criticisms of the NAPs partly reflected distributional welfare concerns about the sheer levels of free allocation given to industry. For example, Trotignon & Delbosc (2008) found evidence of large surplus allocations of allowances relative to emissions to non-combustion sectors of the EU ETS in Phase 1, with allocations ranging in the order 104.2% to 120.3% of actual emissions. Pearson (2010) calculated that even before the economic crisis of 2009, similar levels of surplus allocations continued in Phase 2, with the potential for windfall profits in certain sectors. Studies examining the degree of abatement in the non-electricity sectors have also shown that this level of surplus allocation can almost entirely be attributed to high allocations, rather than significant abatement (Abrell et al, 2011). The latter paper also provided econometric evidence suggesting that, in both 2005 and 2008, firms which were allocated a lower ratio of free allowances compared to their emissions tended to reduce their emissions in the subsequent year by much more than those which received higher free allocation ratios. The results suggest that the level of allocation may have a meaningful impact on abatement levels and can therefore be a source of inefficiency.

Several authors have also focused on the possibility for distortions in final product markets in EU ETS Phases 1 and 2. Free allocations methodologies under the NAPs were also found to be poorly harmonised across Member States. As Member states exercised considerable discretion about allocations to installations on their own territories, the risks of intra-EU competitiveness or investment location distortions were found to be high (Betz et al, 2004 and 2006; Betz and Sato, 2006; del Rio Gonzalez, 2006). These effects were made worse by emissions baseline inflation, as identified empirically by Neuhoff et al (2006) and Anderson and Di Maria (2011). Even within Member States, the combination of grandfathering and windfall profits in the power sector was found empirically to be distortionary for investments in new power plant capacity by Pahle (2011) and theoretically shown to be possible by Golombek (2013). Thus, although carbon leakage risks seemed to have been effectively mitigated – Ellerman et al (2010) and Sartor (2012) found no statistically significant evidence of leakage occurring during EU ETS Phases 1 and 2 – potential for intra-EU competitiveness and abatement cost distortions remained.

Several studies have therefore looked at ways of improving the design of anti-leakage measures. The economic literature generally finds border carbon adjustments to be both the most economically efficient

and environmentally effective way to mitigate carbon leakage risks (cf. Branger and Quirion (2013) for a survey of existing studies). However, given political challenges in implementing border measures, authors have also analysed how best to initially allocate free allowances to mitigate carbon leakage risk (in the absence of border carbon adjustments). Demailly and Quirion (2006), Böhringer et al (2010), Golombek (2013) all find that output-based allocations better mitigate carbon leakage and leave less scope for windfall profits. However, they do so at the significant cost of reducing economic efficiency and potentially the environmental effectiveness of the carbon market by eliminating the channel of low-carbon product substitution. Martin et al (2012) note that, if the goal of free allocation is to mitigate carbon leakage, then the optimal free allocation level should equate the marginal costs of issuing free allowance to the marginal benefits of mitigating carbon leakage at each installation and they present evidence suggesting that the Phase 3 EU ETS allocation rules do not do so as well as they could. Clò (2010) and Dröge & Cooper (2010) analyse the trade-exposure criterion for determining if a sector is at risk of carbon leakage and determine that it unnecessarily inflates the list of sectors considered at risk.

This paper is situated within this latter literature which seeks to evaluate ways in which free allocations, if they are to be used instead of border measures, might be improved. The present article seeks to contribute to the existing literature on optimal anti-leakage policies by showing empirically what an ex-ante benchmarking system looks like in practice from a welfare and efficiency stand-point. In particular, data on the cement sector is used to focus on the extent to which Phase 2 allocations were potentially distortionary to the internal market for energy intensive products and on the extent to which the introduction of harmonised free allocations actually reduce these distortion risks. In addition, the paper presents a series of stylised facts which serve to highlight some of the important ways in which Phase 3 allocation rules have concretely improved the welfare and efficiency properties of free allocations in the EU ETS.

The remainder of the chapter is organised as follows. Section 2.2 describes the essential changes in the free allocation rules which apply in EU ETS Phase 3 compared to Phase 2. Section 2.3 briefly presents stylised facts which highlight some of the main efficiency and distributional welfare implications of the new allocation rules concerning the likelihood of windfall profits, carbon leakage and comparative distributional impacts on Member States. Section 2.4 then shows that while harmonised benchmarks help to reduce the risks of competitiveness distortions in the internal market, the use of ex-ante output levels to determine allocations still leaves considerable scope for distortions when installations are running below full capacity.

## 2.2. The new benchmarking rules

In Phases 1 and 2 of the EU ETS, over 90% of allowances were allocated free of charge to installations, based on a decentralised system whereby each Member State drew up its own national allocation plan and emissions were roughly allocated according to historical emissions or capacity (Ellerman et al, 2010). The basic formula that determines each installation's allocation for each of its eligible products can be summarized as follows (EC, 2011)<sup>6</sup>:

$$FA_{i,p,t} = BM_p \times HAL_{i,p} \times CLEF_{p,t} \times CSCF_t \quad (1)$$

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<sup>6</sup>In some cases benchmarks for specific products cannot easily or practically be used and so hierarchy of fallback approaches is used, based firstly on heat and then fuel consumption benchmarks and, if these are not possible, historical process emissions x 0.97 are used.

$FA_{i,p,t}$  is the total free allocation that installation  $i$  receives for its product  $p$  in year  $t$ .

$BM_p$  is the product emissions-intensity benchmark of product  $p$ . It is typically measured in tones of CO<sub>2</sub>e/unit of output, and is typically based on the average emissions intensity of the 10% most efficient installations in the EU ETS in 2007-08, although fallback approaches exist when this is not technically possible to evaluate or the measure is considered inappropriate as a sectoral benchmark<sup>7</sup>.

$HAL_{i,p}$  is the reference historical activity (production) level of product  $p$  by installation  $i$ , with installations' operators allowed choosing between the highest value of the median annual production levels over 2005-08 or 2009-10.

$CLEF_{p,t}$  is an allocation reduction factor that is applied to a small minority of products that are not considered to be at risk of carbon leakage (cf. EC, 2010a). These products will see their free allocations reduced by a multiplier of 0.8 in 2013 and the multiplier will decline linearly each year to reach 0.3 in 2020.

Finally,  $CSCF_t$  is a uniform, cross-sectoral correction factor that can be applied to ensure that the total free allocation will not exceed the maximum annual amount of free allocation as defined in Article 10a(5) of the ETS directive. It effectively ensures that the level of aggregate free allocation to non-electricity sectors does not rise as a percentage of the total emissions cap over time.

Some additional complexities, such as the treatment of cross-installation-boundary heat flows, waste gas recovery, and electricity consumption, can affect an installation's free allocation level. Each of these effects will be mentioned where relevant in the discussion below.

## **2.3. The empirical evidence on Phase 3 allocations: welfare and efficiency implications of the new rules**

### **2.3.1. Data description**

This analysis uses EU ETS installations compliance data from the European Union Transaction Log (EUTL) for the period 2008-11. These data were matched with the preliminary annual free allocation data for each installation for the period 2013-20 as reported in the National Implementation Measures (NIM) of 20 Member States using installation identifiers that were common to both the NIMs publications and the EUTL data on emissions and historical allocations. The NIMs are the basic document which describes the preliminary free allocation proposed to each installation during Phase 3, prior to validation by the European Commission and the application of the cross-sectoral correction factor. Each installation was then further matched with a 4 digit-level NACE code to identify its primary production activity. This matching was done using a list of EU ETS installations and their NACE codes provided on DG Competition's website (DG Competition, 2009). Missing Member States are Belgium, Hungary, Malta, Lithuania, Slovenia, Czech Republic and Latvia, whose NIMs were not available or not in a match-able form at the time of writing.

Since the changes in allocation levels to new entrants in Phase 3 were not able to be calculated, this paper ignores the effects of the benchmarking rules on new entrants. Further excluding new entrants,

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<sup>7</sup> Where the best 10% of installations emissions intensity could not be gauged, fallback approaches were used based on best available technology literature.

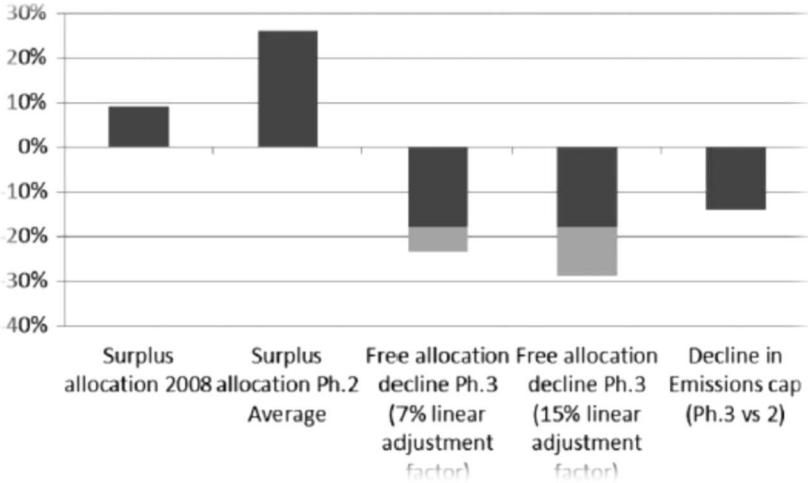
the aviation sector, installations which had left the EU ETS in Phase 3, and installations which could not be matched with either a EUTL installation code or a NACE code, left a sample of 7149 installations which together accounted for 1.46 billion tones of CO2 or approximately 80% of EU ETS emissions in 2010 (EUTL, 2011). Of these, 4174 installations were identified by their NACE code as non-electricity installations and thus subject directly to benchmarking. From those 4174 installations, 329 specializing in the chemicals and non-ferrous metals sector were excluded from the analysis since these sectors have had their EU ETS perimeter change significantly between Phases 2 and 3 and hence changes in allocation could not be attributed to benchmarking alone.

**2.3.2. For benchmarked sectors, Phase 3 free allocation levels will fall significantly compared to Phase 2.**

Perhaps the most striking feature of the collected data on allocations in Phase 3 is that it shows that free allocations to benchmarked sectors will fall significantly in Phase 3. For our sample of 3845 benchmarked installations, the aggregate decline in free allocation compared to Phase 2 will be 20.6% on average. For installations with at least 90% of their products considered exposed to carbon leakage risk (3102 of the 3845 installations in the final data sample), the decline is slightly lower at 17.6%. Note that this is the decline in allocation before taking account of the application of the uniform linear adjustment factor referred to in equation (1). The adjustment factor, which was yet to be announced at the time of writing, has been variously estimated at between 93% (Lecourt, 2013) and 85% (Graichen et al, 2013) of the preliminary allocated amount. These estimates would therefore a total decline in allocations in Phase 3 of 24.2 to 32.6% compared to Phase 2 levels.

Figure 2.1 summarises these results for the 3102 “carbon leakage-exposed” installations and compares them to the aggregate surplus allocation experienced by the same 3102 installations in 2008 and over Phase 2 as a whole. The results show that the decline in allocations from simply moving from grandfathering to benchmarks more than offsets the aggregate surplus allocation in 2008 (prior to the effect of the economic downturn). Meanwhile, assuming a linear reduction factor of between 7 and 15% in addition leads to the total decline in free allocation in Phase 3 more of less fully offsetting the size of the allowance surplus in Phase 2 on average (which was mostly due to the economic downturn).

**Figure 2.1 Estimated net CO2 costs for the median installation in each sector as a percentage of sector gross value added at 25€/tCO2 and with a -15% correction factor**



Sectors included in this analysis: Pulp and paper, steel, coke, refining, cement, lime, ceramics, glass, and ferrous metals production.

The declines in free allocation are also quite uniform across the key sectors affected by benchmarking (Table 2.1). With the exception of what we define here as “other sectors” (which includes a large number of sub-sectors not deemed exposed to carbon leakage and therefore facing a larger reduction factor on average) all of the declines fall in the relatively narrow range of -13 to -24% prior to the linear reduction factor. Some sectors will see their allocation changes affected more or less than others by special rules that have greater impacts on their specific sector<sup>8</sup>.

**Table 2.1 Percentage change in allocation per sector**

	Cokery	Refined petrol products	Glass	Ceramics and brick	Cement	Lime	Pulp and paper	Iron and steel	Other sectors
Aggregate net position 2008 <sup>a</sup>	-16	-1	+9	+35	+11	+15	+22	+29	+13
Aggregate allocation change <sup>b</sup>	-17	-24	-24	-16	-13	-19	-21	-13	-37
Median allocation change <sup>c</sup>	-6	-14	-21	-17	-11	-18	-22	-11	-33
Dispersion of allocation changes <sup>d</sup>	25	26	19	28	9	18	1331	82	99

<sup>a</sup>Calculated as (allocation – emissions)/emissions in 2008 in the sector.

<sup>b</sup>Refers to the total aggregate reduction in allocation in the sector.

<sup>c</sup>Refers to the median reduction in allocation of installations in the sector.

<sup>d</sup>Refers to the average distance from the median allocation reduction of installations in the sector.

### 2.3.3. Welfare implications: lower risk of windfall profits, minimal carbon leakage risks and redistributions within Member States

The generalised decline in free allocations in Phase 3 compared to Phase 2 is likely to have several important consequences for both the actual functioning and perceptions of the EU ETS. From an welfare distribution perspective, to the extent that allocations well in excess of emissions are likely to be curtailed, risks of windfall profits occurring in the non-power sectors are likely to be much lower than during Phases 1 and 2.

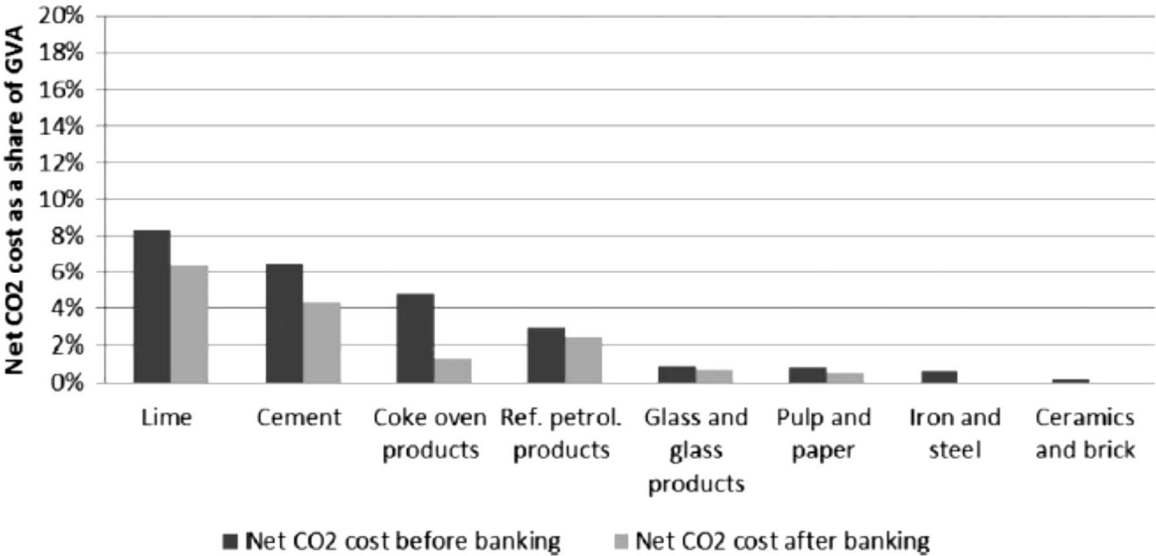
Of course, even at allocation levels below 100% of historical emissions, windfall profits could still potentially occur if, despite majority free allocation, carbon prices were still passed through to final consumers in marginal product prices in these sectors. However the empirical evidence to date has yet to confirm that this occurs in the sectors concerned. This would also require that international competition is weak enough to allow for domestic EU prices to rise by enough to include the opportunity cost of carbon in the concerned sectors. Another, more realistic, alternative by which windfall profits

<sup>8</sup> For example, despite generally large allocations in 2008 relative to emissions, the steel sector only sees a reduction of -13% on aggregate in Phase 3. This reflects the fact that, under benchmarking, a significant share of emissions allowances is allocated to steel sector installations for waste gas emissions that occur offsite. The cost of the emissions from these waste gases however affect the electricity price steel makers receive for the waste gases that are provided to electricity generators, who provide electricity in return. Similarly, the refined petroleum sector sees a larger decline (-24%) despite its deficit in 2008 because of its large share of electricity auto-production, which is not compensated via the benchmarking mechanism but via other mechanisms, such as direct state aid.

could still occur under the new benchmarking rules would be if installations were to produce at levels significantly below the ex-ante historical activity level used to calculate their free allocations. As will be shown in Section 4.4, this remains a significant possibility in some sectors given the effects of the economic down-turn and the linking of allocations to ex-ante historical activity level.

Despite the fact that free allocations have been significantly reduced, it does not appear to be the case that this will create a significant risk of carbon leakage. This is because the free allocations to the main sectors exposed to carbon leakage risk will have sufficiently large allocations to mitigate all but a small minority of their net compliance costs. Figure 2.2 shows this. It is calculated based on the most extreme conditions, in terms of compliance costs, that could be expected to obtain in Phase 3 of the EU ETS. Specifically, the calculation assumes a carbon price of 25€/tCO<sub>2</sub>, that installations produce at their historical activity levels through-out Phase 3 and a cross-sectoral correction factor on NIM allocations of -15%. It is also assumed that firms have no ability to pass-through costs, abate emissions or use cheaper carbon offsets.

**Figure 2.2 Net Cost of CO<sub>2</sub> as a share of sectoral GVA**



Notes: Figure shows estimated net CO<sub>2</sub> costs for the median installation in each sector as a percentage of sector GVA at 25E/tCO<sub>2</sub> and with a -15% correction factor. Data on GVA for the EU27 come from the European Commission’s Impact Assessment Report of 2009 (EC, 2009b). Where a sector has several relevant products with different carbon cost values, the average CO<sub>2</sub> cost impact was taken. Calculations assume no possibility for abatement, carbon price pass-through, or carbon offset use to reduce costs. Indirect carbon costs from electricity prices are ignored. Estimates of banked allowances are based on EUTL data of verified emissions vs. Phase 2 allocations and assume no sales or purchases of allowances

Even with these assumptions only the lime and cement sectors, for which notoriously high transport costs are a significant barrier to carbon leakage, are estimated to have post-allocation costs of more than 4% of their gross value added. Moreover, after the effects of banked allowances from Phase 2 are taken into account the cost estimates are further reduced in every sector.

For the purposes of this calculation only real costs – and not opportunity costs – are taken into account. Marginal pricing theory suggests that opportunity costs could still lead to carbon leakage, since profit maximizing firms should include opportunity costs in their pricing and production volume decisions. However, evidence to date suggests that despite carbon prices of 20 - 30€/tCO<sub>2</sub> in 2005-2006 and 2008,



operational carbon leakage does appear to have occurred at any detectable scale (cf. Branger & Quirion, 2013; Ellerman et al Ch.8, 2010; Reinaud, 2008; Sartor, 2012). This result implies that firms are either not pricing opportunity costs into their product prices or, if they are, they are not leading to detectable levels of carbon leakage at these prices. It therefore seems reasonable to conclude that the new allocation rules therefore are likely to be sufficient to mitigate leakage risks in these sectors, even as they reduce free allocations significantly compared to Phase 2.

Another interesting distributional implication of the new allocation levels is that they do not appear to strongly redistribute free allowances from certain countries in the EU to others. The redistributive effects of common EU-wide benchmarks are much more important at the intra-Member State level than the inter-Member State level. Table 2.2 illustrates this point. It decomposes the variance of inter-Phase allocation changes in two parts. The first part consists of the variance between each installation and its national average allocation decline, while the second part represents the remaining variance between each installation and the sectoral average allocation change. The results show that for the key sectors some redistribution of allowances will occur across Member States, but this redistribution is generally small compared to redistributions within Member States. The use of common, EU-wide emissions performance benchmarks therefore does not appear to induce significant levels of inter-country transfers of allowances compared to EU ETS Phase 2. Consequently, the rules do seem to be disproportionately disadvantaging carbon intensive industry in certain Member States by redistributing allowances away from them to other “better technology” Member States.

**Table 2.2 Decomposition of the installation allocation change variance of benchmarked sectors**

	Pulp and paper	Cokery	Refined petroleum products	Glass	Ceramics and bricks	Cement	Lime	Iron and steel
Inter-country	11%	44%	27%	9%	16%	26%	26%	10%
Intra-country	89%	56%	73%	91%	84%	74%	74%	90%

**2.4. Has the potential for allocation-related distortions in the EU’s internal market been eliminated by harmonised allocation rules?**

Section 1 noted that one of the primary criticisms of the decentralized free allocations in Phase 2 was that it allowed for potential competitiveness distortions within the EU’s internal market because of differing levels of allocation to installations in the same sector. This section therefore asks: to what extent has the potential for competitive distortions of the internal market been reduced by moving from a decentralized to a centralized allocation system? The discussion proceeds in two steps. First, an econometric analysis is presented which demonstrates that the more flexible and decentralised approach to free allocations in Phase 2 led to a significant degree of unexplained heterogeneity in final free allocation rates for installations and Member States competing in the same sectors. Secondly, it is shown that while these potentially distortionary differences are now eliminated by using harmonized benchmarks, two other important sources of potential market distortions remain: the use of ex-ante output levels and the use of output-thresholds to determine allocations.

### 2.4.1. Modelling approach and data

Phase 3's product benchmark-based free allocation rules are intended to provide a greater level of harmonisation of the free allocation rules and the actual levels across installations producing like products. But while Phase 3 benchmarking rules are harmonized almost by definition, the marginal benefit of this formal harmonisation is not immediately obvious. Rather, it depends on the extent to which the prior system was not harmonised and the potential for competitiveness distortions created by the previous lack of harmonisation. The modeling approach developed below therefore attacks this question by first presenting evidence of the extent to which Phase 2 free allocations could be said to be poorly harmonised both across and within Member States. It is shown that the size of the unexplained heterogeneity in allocation levels to installations in Phase 2 was large enough on a gross-margin basis to have been distortionary to primary product markets in at least one sector. For tractability, the analysis here is restricted to the cement sector. The cement sector was chosen because, among all the sectors, it has the most homogeneous production process as far as emissions are concerned (clinker production). Moreover, to the best of our knowledge, EU ETS cement sector installations do not commonly have significant cross-boundary heat flows which could meaningfully affect their free allocation levels in Phase 3 due to the new rules.

To begin with, we specify the following econometric model of annual allocation changes between Phase 2 and Phase 3 in the cement sector:

$$\Delta ALLOC_i = \alpha + \beta_1 NetPos2008_i + \beta_2 CO2Intensity_i + \sum_{j=2}^N \beta_j country_{ij} + u_i \quad (2)$$

$\Delta ALLOC_{ij}$  is the percentage change in the (average) annual free allocation level of cement installation  $i$  in moving from Phase 2 to Phase 3. It is based on the average allocation level per year in Phase 2 versus that of Phase 3.  $NetPos08_i$  is a measure for the extent to which installation  $i$  was either over- or under-allocated relative to its verified emissions in Phase 2. Specifically, it is the ratio of installation  $i$ 's aggregate free allocation in 2008 divided by its verified emissions in the same year. As above, the year 2008 is chosen to control for the effects of the severe drop in industrial production in 2009 and thereafter due to economic conditions. Meanwhile,  $CO2Intensity_i$  is a measure of the CO2 intensity of installation  $i$ . This variable is calculated by first identifying the historical activity level year, based on emissions levels in each of the candidate years reported in the EUTL. This figure is then divided by the average annual free allocation level which firms will receive in Phase 3 according to the NIMs. Since the implied production levels of the emissions and allocations are for the same year for the nominator and denominator, the result gives a direct measure of the ratio of the actual emissions intensity factor of the installation to the cement benchmark that applies to it. Country dummy variables are included in the regression to observe country specific effects. While  $u_i$  represents variation in allocations due to unobserved factors and  $\alpha$  is a constant term.

The logic underlying the model estimation is as follows. To begin with, it is assumed that free allocations to the cement sector in Phase 2 were based on a pure grandfathering approach, i.e. free allocations to installations should have depended only on historical emissions, as indeed has been suggested was the case by Ellerman et al (2010). Now, if this were true then this system could indeed be considered a well-harmonised free allocation method. After all, such a system would imply that there would be no possibility of systematic allowances surpluses or deficits occurring in some Member States but not others.

If such a harmonised grandfathering approach was indeed the practice in Phase 2, this would imply that, as the EU moved to an EU-wide harmonised benchmark-based allocation, the key determinant of the size of the declines in free allocation to each installation must be the difference between historical emissions intensity and the new emissions intensity benchmark. Thus, in terms of Equation (2), only  $CO2Intensity_{ij}$  should be statistically significant, but not the other variables.

Alternatively, if, say, Phase 2 surplus allocations (proxied in Equation (2) by  $NetPos2008_{ij}$ ) were found to be statistically significant, thus explaining the change in allocations from Phase 2 to Phase 3, this would imply that Phase 2 allocations were not solely based on historical emissions. Rather, it would mean that some additional unexplained heterogeneous<sup>9</sup> level of allocation was given to different installations during Phase 2 that was unrelated to its emissions performance or its historical output level<sup>10</sup> and that this was being “corrected” for by the imposition of the new harmonized EU-wide rules.

### 2.4.2. Installation-level OLS regression results for the cement sector

To estimate the model we used 223 observations on the cement sector from the already described database. To remove outliers resulting from capacity changes from biasing the results, the sample was reduced to omit a small number of installations with allocation increases of 50% of their maximum and installations which had seen drops of over 90% in their allocations. Table 2.3 describes the key features of the remaining data:

**Table 2.3 Regression data summary**

	Allocation Change	Net Position 2008	CO2 Intensity (Actual/BM)
Mean	-10.3%	10.9%	94.3%
Median	-11.4%	8.0%	91.9%
St. Dev.	11.9%	19.4%	11.5%
Max	41.2%	121.6%	153.2%
Min	-52.2%	-32.0%	65.0%
N	223	223	223

Post-estimation analysis (Shapiro-Wilk test) showed that the estimated residuals were approximately normally distributed, but Breusch-Pagan tests indicated some (weak) evidence of heteroskedasticity (cf.

<sup>9</sup> If the additional free allocation were homogeneous across Member States then it would be captured by the constant and sector dummies.

<sup>10</sup> Note that since the dependent variable is determined based on the historical activity level used in the benchmark calculation, historical activity is controlled for implicitly in the model.

Appendix). Robust standard errors were therefore used for significance testing. A scatter plot showed no signs of outlier-driven bias in the final sample (cf. Figure 2.3).

**Figure 2.3 Net Position 2008 vs. Change in free allocation in Ph3 vs. Ph2.**

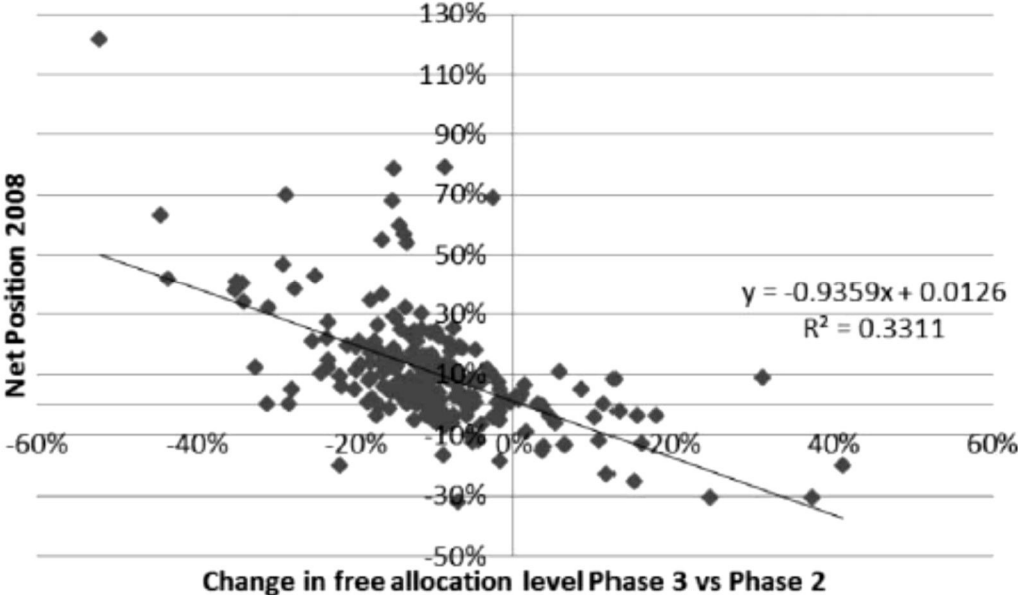


Table 2.4 presents the results of the regression estimation, which was performed using Ordinary Least Squares. Under all three alternative specifications the  $NetPos2008_i$  coefficient was found to be consistently negative and statistically significant at the 99% confidence level. The negative sign implies that, on average, the higher (lower) was an installation’s over- (under-) allocation compared to its historical emissions, the greater (smaller) is that installation’s reduction in free allocation now that it falls under the harmonised rules. These results therefore suggest that a high degree of heterogeneity in free allocation levels existed in the Phase 2 NAPs, even after accounting for different levels of allocation due to historical output and emissions intensity of each free allocation.

**Table 2.4 Installation level regressions results (coefficient estimates for each variable for 4 different Ordinary Least Squares specifications)**

Variable	OLS1	OLS2	OLS3	OLS4
<i>NetPos2008</i>	-0.354 <sup>a</sup>	-0.376 <sup>a</sup>	-0.325 <sup>a</sup>	-
<i>CO2Intensity</i>		-0.467 <sup>a</sup>	-0.464 <sup>a</sup>	-0.400 <sup>a</sup>
<i>MS1</i>				-0.123 <sup>a</sup>
<i>MS2</i>				-0.077 <sup>b</sup>
<i>MS3</i>				-0.166 <sup>a</sup>
<i>MS4</i>				-0.312 <sup>a</sup>
<i>MS5</i>				-0.172 <sup>a</sup>
<i>MS6</i>				-0.137 <sup>a</sup>
<i>MS7</i>				-0.118 <sup>a</sup>
<i>MS8</i>				-0.150 <sup>a</sup>
<i>MS9</i>				-0.190 <sup>a</sup>
<i>MS10</i>				-0.200 <sup>a</sup>
<i>MS11</i>				-0.156 <sup>a</sup>
<i>MS12</i>				-0.131 <sup>a</sup>
<i>MS13</i>				-0.341 <sup>a</sup>
<i>MS14</i>				-0.180 <sup>a</sup>
<i>MS15</i>				-0.250 <sup>a</sup>
<i>MS16</i>				-0.265 <sup>a</sup>
	None	None	Yes	Yes
	-0.065 <sup>a</sup>	0.439 <sup>a</sup>	0.383 <sup>a</sup>	-0.516 <sup>a</sup>
Descriptive Statistics				

R <sup>2</sup>	0.33	0.53	0.67	0.42
Prob > F	0.000	0.000	0.000	0.000
Observations	223	223	223	223

<sup>a</sup>Statistically significant at 2.5% level, <sup>b</sup>Statistically significant at 5% level <sup>c</sup>Statistically significant at 10% level

Standard errors are Newey-West Standard Errors.

Moreover, the results strongly indicate that this heterogeneity accounted for a large share of the final Phase 2 allocation levels to each installation. The coefficient estimates on *CO2Intensity* and *NetPos2008* presented in Table 5 imply that, on average, for every 10 percentage points of decline in free allocation due to the move to stricter benchmarks roughly 4.5 percentage points are due to the fact of having stricter benchmarks than the actual emissions intensity, while approximately 3.5 additional percentage points are effectively correcting for other unexplained sources of differences between the historical emissions level and the historical allocation level in Phase 2. The importance of these other unexplained sources of different allocation levels is further highlighted by the fact that they appear to explain approximately one third of the variation in the dependent variable, as seen by the R-squared of 0.33 in the first specification. This is strong evidence that the lack of formally harmonised allocation rules in Phase 2 leads to a high degree of variation in allocation levels between installations and that this cannot be explained by their actual emissions levels. It thus provides empirical and quantitative proof of the extent to which the concerns that have been raised in NAP literature were in fact valid and were large in magnitude.

Furthermore, the result does not appear to depend on just one or two countries who over or under-allocated their installations more than others. In specification number 4, the *NetPos2008* variable is dropped from the regression and country dummies are added in its place. This specification now captures the average country-specific contribution to the allocation change of each installation, after controlling for its distance from the benchmark level of free allocation and historical output in the same manner as before. Each Member State dummy variable now captures the country-specific level of allocation in Phase 2 that is being adjusted for in Phase 3, relative to Austria, which is the base case and is represented by the constant term.

The results of specification 4 indicate that all 17 Member State-dummies (including the constant) were found to be statistically significant. The country dummy coefficients are widely different from each other, varying from -0.075 up to -0.341, with a median of 0.17. The standard deviation of the coefficients themselves, 0.073, is also well outside the average 95% confidence interval estimate for all of these coefficients, which is plus or minus 0.033. These results confirm that the previous result was not just driven by one or two countries, but by an EU-wide phenomenon.

### 2.4.3. What is the scope for market distortions in the EU cement sector?

It is instructive to give a quantitative indication of the risk of distortions on the primary product markets with such differing levels of allocations in Phase 2 could have created. To do so, it is first assumed that

only residual CO2 emissions costs after deducting the free allocation compensation are incorporated into the unit price of cement by producers, in keeping with the idea of a threat of international competition if prices rise to reflect CO2 opportunity costs. Symmetrically, it is assumed excess free allocations over emissions are considered by firms as a per unit subsidy, i.e.

$$\text{Unit CO2 cost} = \text{EUA price} \times (\text{emissions per unit} - \text{free allocation per unit}) \quad (3)$$

Secondly, the *NetPos2008* coefficient estimate from regression 3 in Table 2.4 is exploited to obtain fitted estimates of the part of the free allocation of each installation (as a share of its emissions) which is due to the unexplained heterogeneity identified earlier. Thirdly, since the econometric analysis in the preceding section was centered on 2008 data, 2008 data on cement sector emissions per unit of output are obtained from the World Cement Sustainability Initiative GNR database and are assumed to be 0.65 tCO2/t cement. Finally, representative Portland cement prices of 75 and 95€/t are used.

**Table 2.5. Effective per-unit subsidy rates to cement installations due to free allocation in Phase 2 (by quintile)**

	Per unit subsidy (€/tcement)	% of output price (low demand)	% of output price (high demand)
5th Quintile	-3,95	-5,3%	-4,2%
4th Quintile	0,37	0,5%	0,4%
3rd Quintile	0,96	1,3%	1,0%
2nd Quintile	1,38	1,8%	1,5%
1st Quintile	2,05	2,7%	2,2%
0th Quintile	11,28	15,0%	11,9%

Based on these data, Table 2.5 presents five different estimates of the effective subsidy (or cost) rate that are estimated to have been received by the different installations. Each estimate corresponds to a different quintile of the distribution of allocation levels observed in 2008 and provides the associated subsidy (or cost) level at carbon prices of 25€/tCO2 (average 2008 prices). The size of the subsidy relative to the two different possible assumptions about output prices are provided to give an indication of the possibility for distortions. The results show that the size of the subsidies in the zeroth and first quintiles are significant as a share of product prices, particularly in the low demand scenario. Note that while the effective subsidy as a percentage of the final product price is only between 2 and 3 percent for the first quintile, 2 to 3 percent of final product prices translate to a much higher share of net margins. Moreover, there is a particularly wide range of effective subsidy rates between the “zeroth”, first and fifth quintiles – particularly in the low demand scenario, with the “effective free allocation subsidy”

ranging from -5.3 to 15% of output prices. This estimation from the cement sector seem to indicate potential for primary market distortions exist and that the move to Phase 3 benchmarks are an important development, at least for sectors with similar cost structures to cement.

**2.4.4. Ex-ante output- data distortions: the case of Greece, Italy, Ireland, Spain and Portugal**

The foregoing discussion on the potential for distortions in EU ETS Phase 2 has thus far ignored an important caveat. Namely, although under benchmarking the product benchmarks applied to the allocation given to each installation are the same, the historical output or activity levels (HAL in equation (1)) are not the same for every installation. For that matter, will be capacity utilisation rate, and thus the level of production relative to the HAL will be different.

**Figure 2.4. Greek, Italian, Irish, Spanish and Portuguese cement sector installations’ 2012 emissions vs. 2013 allocations and historical activity level emissions**

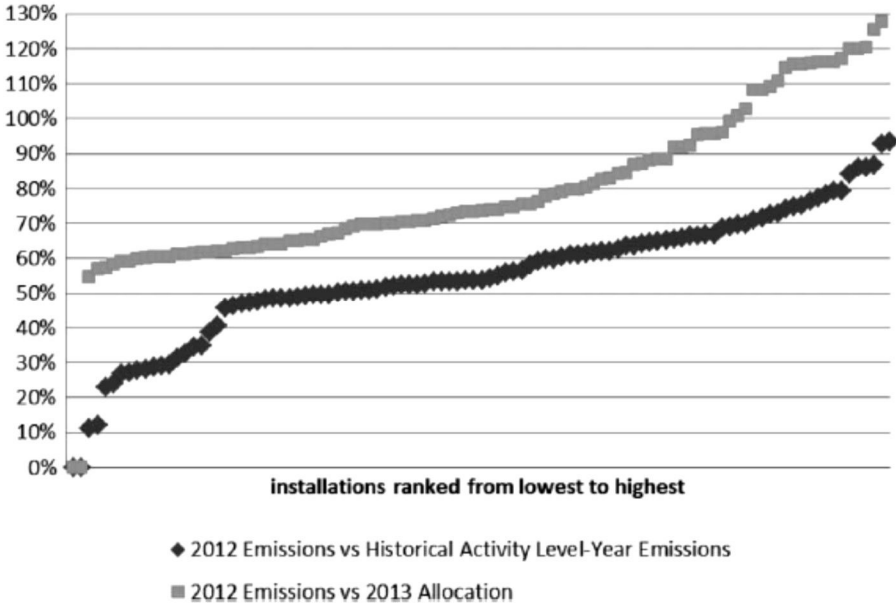


Figure 2.4 shows two comparisons using the example of Greek, Italian, Irish, Spanish and Portuguese cement installations in the EU ETS. First, it compares the 2012 emissions and 2013 allocations of these installations (including the application of a 7% cross sectoral correction factor, close to the 5.7% which will actually apply in that year). It is clear that if emissions levels in 2013 are similar to those of 2012 then the majority of installations would receive a very large surplus allocation of allowances compared to their emissions. This is because 2013 allocations are based on historical output levels that relate to periods before the global financial crisis and the severe collapse in demand that ensued. This is illustrated by the second data series presented in Figure 4, which shows the ratio of the emissions of these installations in 2012 to their emissions in the historical activity level year to determine its Phase 3 free allocations.

Based on the analysis of effective subsidy rates presented earlier it is clear that significant potential still remains for internal EU-market distortions to arise in the cement or similar sectors because of these very



different levels of emissions relative to free allocations. The combination of a formula based on distant ex-ante production levels and a severe sectoral downturn have created similar-sized risks of distortions in the internal market for cement and clinker.

## **2.5. Chapter Conclusion**

The preceding discussion has shown that Phase 3 allocation rules have a number of desirable distributional welfare characteristics, especially compared to those of Phase 2 allocations as identified in the NAP literature. Most notably, they have reduced the potential for windfall profits to accrue to industry by significantly reducing the amounts of free allocations received by industry in Phase 3, which were typically well in excess of their emissions in Phase 2. At the same time, this does not appear to occur at the expense of raising risks of carbon leakage because the residual costs of compliance for these industries will remain small as a share of gross value added (and compared to transport costs in some sectors). Evidence strongly suggests that the introduction of benchmarking does not lead to significant redistributions of allowances from industry in certain member states to others. These are all desirable properties of a free allocation mechanism, although, as noted by Martin et al (2012) among others, they are not necessarily optimal.

On the other hand, although harmonisation of allocation rules has removed one source of potential competitiveness distortions identified by the NAP literature – that due to different allocation rules in different Member States – other kinds of competitiveness distortions are still possible due to the use of ex-ante output data to determine allocations in a time of low capacity utilisation. It may well be the case that this effect is exacerbated by the use of specific output thresholds as part of the new benchmarking rules. These results call for further research and investigation of whether this aspect of the new benchmarking rules does create significant distortions in markets such as those for clinker and cement and how this may impact the efficiency and effectiveness of the carbon market itself.

## 2.6. Chapter Appendix

### 2.6.1. Results of pre-estimation econometric tests

Table 2.6. Shapiro-Wilk Test for normally-distributed residuals:

**H<sub>0</sub>: Residuals are normally distributed. Test:**

Variable	Observations	w	V	Z	Prob>z
Residuals	223	0.97053	1.187	0.362	0.35875

Conclusion: Insufficient evidence to reject H<sub>0</sub>

Table 2.7. Breusch-Pagan Test for heteroskedasticity

**H<sub>0</sub>:  $\sigma(i)^2 = \sigma^2$  for all i (i.e. there is no heteroskedasticity in errors). Test:**

X <sup>2</sup> (3)	5.14
Prob> X <sup>2</sup> (3)	0.1617

Conclusion: Insufficient evidence to reject H<sub>0</sub>

## Chapter 3

# EU ETS, Free Allocations and Activity Level Thresholds<sup>11</sup>

### 3.1. Chapter Introduction

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 paper 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* 2014). 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

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<sup>11</sup> This chapter was co-authored with F.Branger, then at CIRED, J.P.Ponssard, Ecole Polytechnique, M.Sato, LSE Grantham. It has since been published in the Journal of the Association of Environmental and Resource Economists September 2015 as: Branger, Frédéric, Ponssard, Jean-Pierre, Sartor, Oliver and Sato, Misato (2015) EU ETS, free allocations, and activity level thresholds: the devil lies in the details. Journal of the Association of Environmental and Resource Economists, 2 (3). pp. 401-437. ISSN 2333-5955 DOI: 10.1086/682343

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<sup>12</sup> of an allocation and subsequent lump-sum transfer based on historic 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).<sup>13</sup>

Qualitatively, ETS schemes with ALTs approximate OBA: the amount of free allocations will vary with the activity level and the over allocation profits<sup>14</sup> associated with *ex-ante* schemes will be reduced.<sup>15</sup> 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 paper. A recent study on the EU ETS impacts on the cement sector 2005-2013 (Neuhoff et al., 2014)<sup>16</sup> 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 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 paper, such strategic adjustments of output motivated by ALTs is termed “gaming” behaviour, in line with the management literature (e.g. Jensen, 2003). 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:

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<sup>12</sup> 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 to be allocated.

<sup>13</sup> New entrant provision and closing rules were already in place in Phases 1 and 2 of the EU-ETS. A closure rule is also used in the Californian ETS.

<sup>14</sup> Over allocation 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 (Branger and Quirion 2015). Over allocation 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<sub>2</sub> opportunity cost to product prices, despite receiving the allowances for free. Over allocation profits can occur even in the absence of cost pass through, if output fall short of historic levels.

<sup>15</sup> Windfall and over allocation 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, and Sandbag 2011).

<sup>16</sup> Three co-authors of this paper participated in this study and in conducting interviews that were carried out.

- 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<sup>17</sup> 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 paper 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.

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) and 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.* (2014) 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.* (Forthcoming), 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 paper could be potentially relevant to other EITEs with similar characteristics. Altogether, we argue that the benefits of implementing ALTs in terms of reduced over allocation 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. We discuss some broader questions if such a change were adopted.

The paper is organized as follows. Section 2 discusses the EU ETS Phase 3 allocation rules, the predicted gaming behaviour from thresholds and the alternative allocation rules. Section 3 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 4 presents the results. Section 5 concludes and discusses policy recommendations.

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<sup>17</sup> McKinsey (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.

## 3.2. ETS free allocation rules and gaming of ALTs

### 3.2.1. 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”<sup>18</sup>) 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 1 and 2 below. The amount of free allocations to an installation,  $i$ , at period  $t+1$ , for an eligible product,  $p$  is denoted  $A_{i,p,t+1}$ .

$$A_{i,p,t+1} = CSCF_{t+1} \times B_p \times HAL_{i,p} \times ALCF_{t+1} \left( \frac{q_t}{HAL_{i,p}} \right), \quad (1)$$

In equation (1)  $CSCF_{t+1}$  is the uniform cross-sectoral correction factor<sup>19</sup>,  $B_p$  is the benchmark for product  $p$ ,<sup>20</sup>  $HAL_{i,p}$  represents the historical activity level; and  $ALCF$  is the activity level correction factor, which depends on the ratio  $q_{i,p,t}/HAL_{i,p}$ ,  $q_{i,p,t}$  being the output of the eligible product in year  $t$ . The  $ALCF$  defines a step wise function for the thresholds. It is defined as:

$$ALCF_{t+1} \left( \frac{q_t}{HAL} \right) = \begin{cases} 1, & q_t \geq 0.5 HAL \\ 0.5, & 0.25 HAL \leq q_t < 0.5 HAL \\ 0.25, & 0.10 HAL \leq q_t < 0.25 HAL \\ 0, & 0 HAL \leq q_t < 0.10 HAL \end{cases} \quad (2)$$

For new installations, the historic activity level is replaced by the capacity, to be precisely determined according to the rules.<sup>21</sup>

### 3.2.2. 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

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<sup>18</sup> 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)).

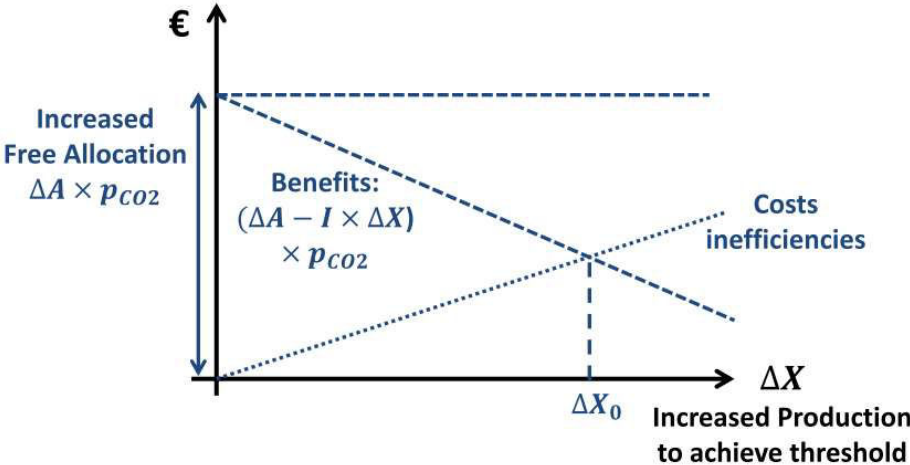
<sup>19</sup> This is determined by comparing the sum of preliminary total annual amounts of emission allowances allocated free to installations (not electricity) for each year over the period 2013-2020. In 2013 the  $CSCF$  is equal to 0.9427, then declines at 1.74% per year.

<sup>20</sup> Product benchmarks in general reflect the average performance of the 10% most efficient installations in the sector or subsector in the years 2007-2008. The benchmarks are calculated for products rather than inputs Decision (2011/278/EU).

<sup>21</sup> Guidance document n°7 in European Commission, 2011.

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 8 €/t CO<sub>2</sub> 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 x 1Mt x 0.766 tCO<sub>2</sub>/t x 8€/tCO<sub>2</sub>, 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 0.8 t CO<sub>2</sub>/t of clinker (slightly above the benchmark). The increase in emissions is then equal to 0.080 t CO<sub>2</sub> which at 8 €/t CO<sub>2</sub> amounts to 0.64 M€.

**Figure 3.1: 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.**



In the presence of activity level thresholds, the net benefit of gaming in terms of allocations 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<sub>2</sub>, the benefit rising with the price. However, this artificial increase of production involves cost inefficiencies, which can be assumed to be increasing function of the extra production, independent of the CO<sub>2</sub> 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 3.1, where gaming is undertaken only if the increased production to attain the threshold is less than  $\Delta X_0$ . 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 (2003) 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.

### 3.2.3. 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 (1) (with  $HAL_{i,p} \times ALCF(q_{i,p,t}/HAL_{i,p})$  replaced by  $q_{i,p,t}$ ). The scheme therefore has no thresholds, and the historic activity level  $HAL$  is replaced by the previous year activity level  $q_t$  so as allocations are altered on a continuous yearly production basis. In this paper, 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.

## 3.3. Methodology and data

### 3.3.1. The cement sector

Our analysis focuses on the cement sector<sup>22</sup> 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.<sup>23</sup> 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; hence 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<sup>24</sup>.

Second, the sector experienced a demand collapse in the order of 50% or more between 2007 and 2012 in several member states. Thus, 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

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<sup>22</sup> For an overview of the European cement sector see Boyer and Ponsard (2013).

<sup>23</sup> It could have been based on a cement or a hybrid benchmark instead. The hybrid benchmark avoids the “clinker-cement paradox” (Quirion 2009).” If the benchmarked product is cement, plants have an incentive to outsource clinker production. If it is clinker, the incentive to reduce the clinker-to-cement ratio is lost. In California, the benchmarked product is “adjusted clinker and mineral additives produced”, which is equal to  $Q_K(1 + \frac{r}{R})$ , where  $Q_K$  is the clinker produced,  $R$  is the clinker ratio and  $r$  is the “mineral additives ratio” (limestone and gypsum consumed divided by cement produced). This system gives an incentive to use more mineral additives while preventing clinker outsourcing.

<sup>24</sup> Some small countries are regrouped into larger entities which are coherent in terms of regional market (see Section C.1).



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<sup>25</sup> with ETS-participating clinker production plants will be divided into two groups (see Table 3.1). The first group includes countries where the average domestic cement consumption in 2011-2012 was less than 70% of 2007 levels.<sup>26</sup> 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 countries represented 51% of EU ETS cement emissions in 2008 and 40% in 2012. The remaining countries are classified as “moderate demand” (MD).

**Table 3.1: Moderate- (MD) and low demand (LD) countries in terms of cement consumption in 2012 relative to 2007 levels<sup>27</sup>**

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

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 ( $q/HAL$ , in the cement sector,  $q$  being the quantity of clinker), but data on output is not publically 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*<sup>28</sup>.

### 3.3.2. 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

<sup>25</sup> 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.

<sup>26</sup> 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.

<sup>27</sup> There are no clinker plants in Malta, Lichtenstein and Iceland. Emissions data on two clinker plants of Cyprus is available from 2012 only, hence cannot be used in this analysis.

<sup>28</sup> 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 (GNR Database). 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 A1, 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 A.2).

relevant market. A simple comparison 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 over allocation 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 characteristics 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<sup>29</sup>) 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 in theory. This is because they are a lump sum transfer and firms would include the opportunity costs of carbon in their marginal costs, even if they are allocated for free. In contrast, with ex-post OBA allocations marginal costs are unchanged because free allocation is directly proportional to output, hence 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 methods are used. This paper 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 sector actors in the EU ETS (Neuhoff et al, 2014, p.26). These interviews point out three reasons why in practice, no price change (cost pass through) was observed in the cement sector so far. First, the ex-ante free allocations were given out, precisely to mitigate carbon leakage. Thus firms perceived a risk of losing future free allocations if they passed through the cost of carbon and there was no leakage. Second, companies reported long term strategic considerations – such as maintaining market share and good client relationships – could partially balance the incentive to pass the carbon price. Third, they perceived the risk of drawing attention of competition authorities due to abnormal profit levels, if the carbon cost pass-through lead to large windfall profits.<sup>30</sup> 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

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<sup>29</sup> For this purpose, we rely heavily on the work carried out by Branger and Quirion (2014), which have developed an installation level dataset for the EU cement sector with clinker producing installations identified.

<sup>30</sup> 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>

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.<sup>31</sup> 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 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 3.4.7, 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<sup>32</sup>, we estimate *the number of free allowances (EUA for EU Allowance, which is the official title 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.<sup>33</sup> 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<sub>2</sub> of additional emissions<sup>34</sup>, 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<sub>2</sub> price at 7.95€/t, which corresponds to the average future price (December 2013) during year 2012<sup>35</sup>. 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 higher or lower price, the important point being that the associated revenue be higher than the associated inefficiency costs (see Section 3.2.2). 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 over allocation profits (in MEAUs or M€). The difference between EXALTG and EXALTNG gives the impact of gaming. Table 2 summarises how allocations and production are obtained under each scenario.

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<sup>31</sup> For estimates see Meunier et al. (2014).

<sup>32</sup> 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).

<sup>33</sup> Caution, in order to make computations easier, this plant does not have the same characteristics as the one in Section 2.2.

<sup>34</sup> Assuming that the plant has a clinker carbon intensity of 800 kg CO<sub>2</sub> per ton of clinker.

<sup>35</sup> Source: ICE database (<http://data.theice.com/MyAccount/Login.aspx>)

**Table 3.2: Scenarios**

<b>Scenarios</b>	<b>Allocations</b>	<b>Production</b>
<i>OBA</i>	Proportional to Activity ( $HAL \times ALCF \leftrightarrow q$ in Eq (1))	Counterfactual (explained in Section C.1)
<i>EX</i>	Independent of Activity ( $ALCF=1$ in Eq (1))	Same as OBA
<i>EXALTNG</i>	Hybrid (Eq (1))	Same as OBA
<i>EXALTG</i>	Same as EXALTNG	Actual 2012 Production

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. Assuming no stockpiling, the remaining part is attributed to the change in the clinker ratio.

### 3.3.3. Estimation strategy

Counterfactual values for clinker plant activity are predicted based on panel data estimations at the plant level. We use first differencing in order to control for country-level time invariant factors and the autoregressive nature of plant activity. The regression includes both country level data (cement consumption, GDP) and plant-level characteristics, such as carbon intensity, size and geographical location (coast) as detailed in Section 3.3.2 and in Appendix 3.7.3. To assess the robustness of our results we use a semi-parametric approach (Powell 1994) by specifically modelling the multiplicative error of our estimation. The counterfactual plant activity level is then not fixed but is a random variable. We perform a Monte Carlo simulation with 10,000 samples and report the average and the 95% confidence interval.

**Table 3.3: Data sources**

Variable	Source
Emissions and HEL	EUTL ( <a href="http://ec.europa.eu/environment/ets/napMgt.do">http://ec.europa.eu/environment/ets/napMgt.do</a> )
Clinker net exports (NE <sub>k</sub> )	Eurostat ( <a href="http://epp.eurostat.ec.europa.eu/newxtweb/setupdimselection.do#">http://epp.eurostat.ec.europa.eu/newxtweb/setupdimselection.do#</a> ) Eurostat ( <a href="http://epp.eurostat.ec.europa.eu/newxtweb/setupdimselection.do#">http://epp.eurostat.ec.europa.eu/newxtweb/setupdimselection.do#</a> , International Trade, EU Trade Since 1988 by HS2, 4, 6 and CN8). Data is originally given by country pairs. Total net exports are re-computed. Product category: “Cement Clinker” (252310)
Cement net exports (NE <sub>c</sub> )	Eurostat Product category: Difference between “Cement, incl. cement clinkers” (2523) and “Cement Clinker” (252310).
Cement consumption (C <sub>c</sub> )	1) Cembureau (2013) for the main European countries 2) VDZ ( <a href="http://www.vdz-online.de/en/publications/factsandfigures/cement-sales-and-consumption/">http://www.vdz-online.de/en/publications/factsandfigures/cement-sales-and-consumption/</a> , Table C10) for Baltic countries and Norway.
Country GDP (GDP)	World Bank Development Indicators Database ( <a href="http://databank.worldbank.org/data/views/variableselection/selectvariables.aspx?source=World-Development-Indicators">http://databank.worldbank.org/data/views/variableselection/selectvariables.aspx?source=World-Development-Indicators</a> ) They are in billion current US dollars
Clinker production (Q <sub>k</sub> )	EUTL-derived estimation (through estimated clinker carbon intensity and emissions, see A1). Where there were data gaps, supplementary data were obtained from several sources e.g.: <ul style="list-style-type: none"> <li>National cement association data when reliable and exploitable, i.e. Spain (<a href="https://www.oficemen.com/Uploads/docs/Anuario%202012%281%29.pdf">https://www.oficemen.com/Uploads/docs/Anuario%202012%281%29.pdf</a>, p90)</li> <li>Germany (<a href="http://www.vdz-online.de/en/publications/factsandfigures/cement-data-at-a-glance/">http://www.vdz-online.de/en/publications/factsandfigures/cement-data-at-a-glance/</a>, table A2)</li> <li>France (<a href="http://www.infociments.fr/publications/industrie-cimentiere/statistiques/st-g08-2012">http://www.infociments.fr/publications/industrie-cimentiere/statistiques/st-g08-2012</a>, Table p7)</li> <li>Getting the Numbers Right database (GNR, <a href="http://wbcsdcement.org/GNR-2012/index.html">http://wbcsdcement.org/GNR-2012/index.html</a><a href="http://wbcsdcement.org/GNR-2012/index.html">http://wbcsdcement.org/GNR-2012/index.html</a>, indicator 311a) for available countries (UK, Italy, Poland, Czech Republic, Austria)</li> </ul>

### 3.4. Results

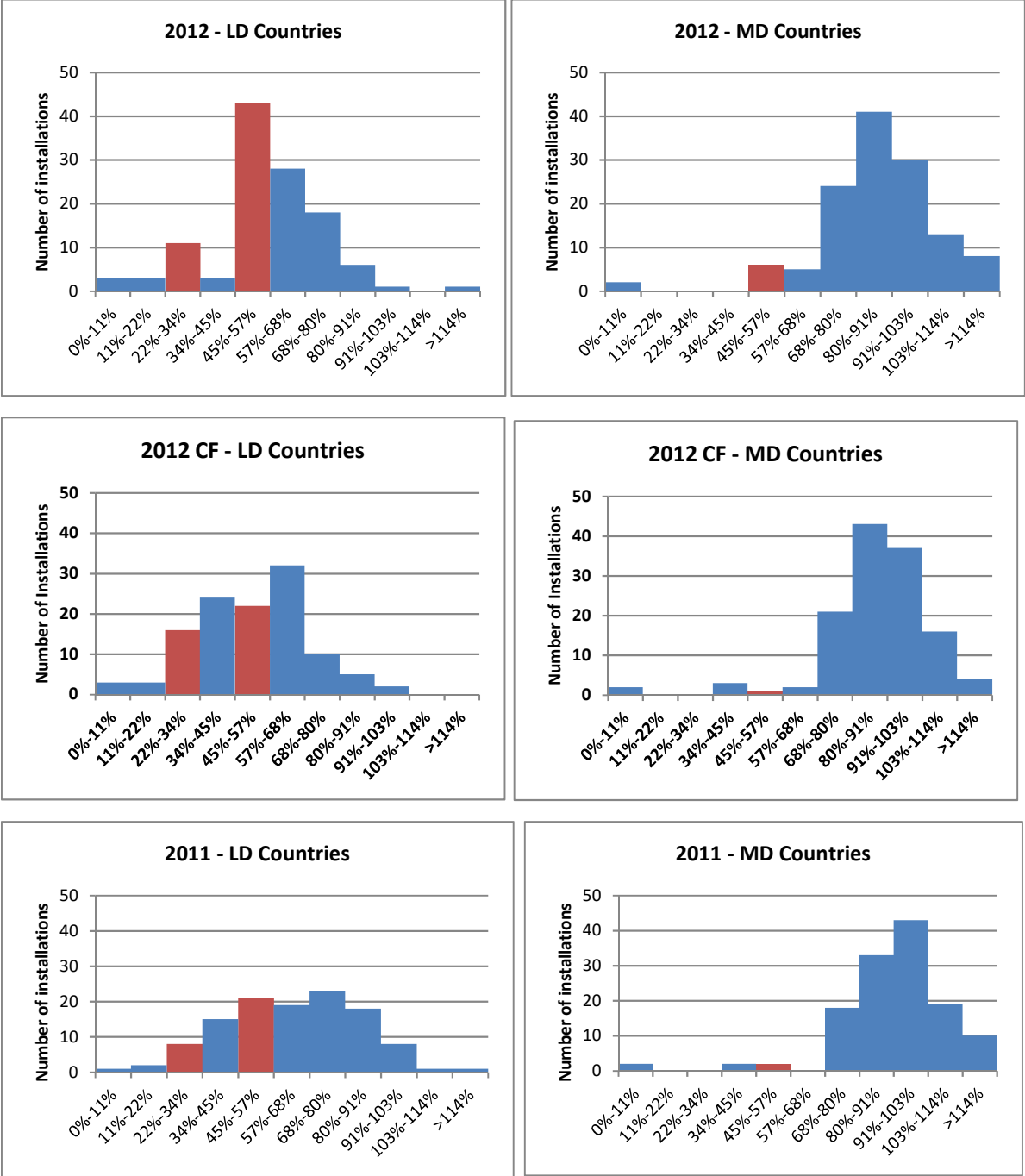
#### 3.4.1. Impact of ALTs on the plant distributions

Figure 3.2 displays the distribution of plant activity levels for 2012 (EXALTG), the counterfactual<sup>36</sup> 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

<sup>36</sup> 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).

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 behaviour.

**Figure 3.2: Distribution of installations according to their activity level (approximated by E/HEL) in 2012 for observed and counterfactual production.**



Notes: 2012CF stands for counterfactual of 2012. Red bars indicate categories just above thresholds. 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 for more explanations).

### 3.4.2. ALTs impacts on clinker production and emissions

Table 3.4 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<sub>2</sub> (+5 % for the sector as a whole) have been emitted by EU cement firms as a consequence of the strategic behaviour of cement companies.

**Table 3.4: Production and Emissions for the observed (EXALTG) and counterfactual (EX, OBA, EXALTNG) scenarios**

	<b>LD countries</b>	<b>MD countries</b>	<b>All countries</b>	<b>Spain</b>	<b>Greece</b>
Production (CF) in Mt	47.2 [45.2,49.4]	80.2 [76.9,83.7]	127.4 [123.6,131.5]	12.4 [11.5,13.5]	3.6 [3.0,4.3]
Production (observed) in Mt	54.4	79.4	133.8	16.0	5.6
<b>Increased Production in Mt</b>	<b>+7.2</b> [5.0,9.2] <b>p=1.00</b>	<b>-0.8</b> [-4.2,2.5] <b>p=0.33</b>	<b>+6.4</b> [2.3,10.2] <b>p=1.00</b>	<b>+3.5</b> [2.5,4.4] <b>p=1.00</b>	<b>+2.0</b> [1.3,2.6] <b>p=1.00</b>
<b>Increased emissions in Mt CO<sub>2</sub></b>	<b>+6.4</b> [4.5,8.2] <b>p=1.00</b>	<b>-0.6</b> [-3.6,2.2] <b>p=0.34</b>	<b>+5.8</b> [2.2,9.1] <b>p=1.00</b>	<b>+3.1</b> [2.2,3.8] <b>p=1.00</b>	<b>+1.8</b> [1.2,2.3] <b>p=1.00</b>

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.

### 3.4.3. Impact of gaming on plant distribution on the free allowances

Table 3.5 gives the amount of EUA's that are allocated to cement installations under the four scenarios (EX, EXALTNG, EXALTG, OBA). If installations received 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.

**Table 3.5: The Free Allowances (MEUAs) under the four scenarios**

Allocations	LD countries	MD countries	All countries	Spain	Greece
EX	74.5	70.0	144.5	23.6	8.7
EXALTNG	55.1 [52.8,57.3]	68.1 [67.2,68.9]	123.2 [120.8,125.6]	14.9 [13.5,16.3]	4.3 [3.5,5.1]
EXALTG (observed)	68.4	69.6	138.1	20.7	7.3
OBA	36.1 [34.5,37.7]	62.2 [59.6,64.9]	98.2 [95.2,101.5]	9.5 [8.7,10.2]	2.7 [2.2,3.2]
Allowances Gaming Gain	+13.3 [11.1,15.6] p=1.00	+1.5 [0.7,2.4] p=1.00	+14.8 [12.5,17.3] p=1.00	+5.8 [4.4,7.2] p=1.00	+3.0 [2.2,3.8] p=1.00
Net Gaming Gain (minus Emissions)	+6.9 [4.9,9.0] p=1.00	+2.1 [-0.5,5.0] p=0.94	+9.0 [5.7,12.5] p=1.00	+2.8 [1.7,3.8] p=1.00	+1.2 [0.6,1.8] 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.

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 over-allocation in 2012 by 6.4 MEUSs compared to the scenario without ALTs. Had OBA been implemented instead, over-allocation 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.

#### 3.4.4. Potential financial 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<sub>2</sub> price. It should be clear that the amount of profitable gaming is dependent on the CO<sub>2</sub> price. We shall come back to this point in our discussion of the results. Table 3.6 gives the potential profit associated with gaming for a CO<sub>2</sub> 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<sub>2</sub> price decreased the following year, but firms may have banked these extra allowances and the CO<sub>2</sub> price may rise in the future).

**Table 3.6 Quantification of the monetary value of excess free allocations for the various scenarios.**

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

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

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<sub>2</sub> 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 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.

### 3.4.5. 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.

**a) 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 3.7 presents four examples<sup>37</sup>. In each of these firms in 2012, production (within the same geographical country) simultaneously falls in one plant (which produced well above the threshold in 2011), and rises in another plant above the threshold (which was previously operating below the threshold).

**Table 3.7: Evidence of within-firm-country production shifting to meet thresholds**

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 A1 for more explanations).

**b) Exports.** Table 3.8 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

<sup>37</sup> 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.

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.

**Table 3.8: Clinker net exports in 2010, 2011 and 2012 in LD and MD countries in Mt**

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.

**c) 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 ton of cement. The clinker ratio can be recomputed at the macro level (state of group of states) with the formula  $R = \frac{Q_K - NE_K}{C_C + NE_C}$ , 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 3.7.2 for explanation and Table 3.3 for data source). Table 9 shows the clinker ratio for the MD countries, LD countries, Spain and Greece. There is some suggestion that the historically declining trend in the clinker-to-cement ratio reversed in 2012, notably in Spain and Greece.

**Table 3.9: Clinker-to-Cement Ratio in selected areas (source: authors' analysis)**

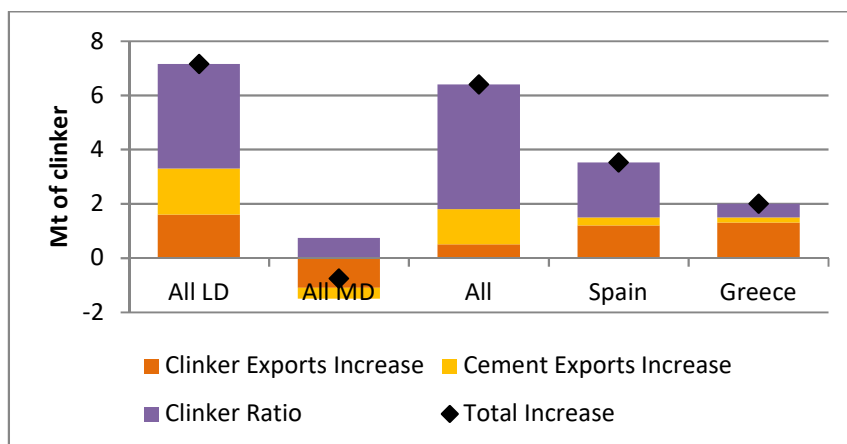
Clinker Ratio	2010	2011	2012
MD Countries	76%	76%	77%
LD Countries	74%	72%	74%
Spain	79%	76%	82%
Greece	76%	71%	75%

### 3.4.6. 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<sup>38</sup> 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 3.7.3.2 for the estimation method and data used). Assuming no stockpiling, we can attribute the remaining excess clinker output to clinker ratio increase. Table 3.10 gives the results. Figure 3.3 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 3.4.6 suggesting that stockpiling of excess clinker output may be occurring, as well as increased clinker ratio of cement exports.

**Figure 3.3: Routes of excess clinker production decomposition**



<sup>38</sup> Production shifting in multiplant companies does not generate excess clinker output, hence is not quantitatively assessed.

**Table 3.10: Real and counterfactual net exports of clinker and cement (Mt)**

Region	Total Increase	2012 Clinker Net Exports			2012 Cement Net Exports			Clinker Ratio	
	Production Clinker	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%

### 3.4.7. Robustness: the impact of hypotheses $H_1$ and $H_2$ on results

In this section we will discuss how results (mainly Tables 3.4 to 3.6) 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 carbon price pass-through in EX/EXALTNG, but not in OBA. Under  $\overline{H_1}$ , **clinker production (and thus emissions) would thus 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 on consumption, using this reduced consumption level instead of the observed one would mean that **under  $\overline{H_2}$ , 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 3.11: Change in main results with  $\overline{H}_1$  and  $\overline{H}_2$**

	Emissions variation EXALTG vs. EXALTNG	Profits <sup>1</sup> EXALTG vs. OBA	Profits <sup>1</sup> EXALTG vs. EXALTNG
	(Extra Pollution due to Gaming)	(Over-allocation Profits)	(Extra Profits due to Gaming)
$\overline{H}_1$ : Pass-through in OBA versus EX/EXALTNG	0	$p_{EUA} \times$ $[\delta E_1 - \delta A_1^{OBA}]$ $(\geq 0)$	0
$\overline{H}_2$ : Some of th surplus of production due to gaming poured into local market	$+\delta E_2$	$p_{EUA} \times$ $[\delta A_2^{OBA} - \delta E_2]$ $(\lesssim 0)$	$p_{EUA} \times$ $[\delta A_2^T - \delta E_2]$

<sup>1</sup> 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 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 3.11 sums up how three main results are modified with  $\overline{H}_1$  and  $\overline{H}_2$ : extra pollution due to gaming, over-allocation 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 flooded the local market driving down the prices). Second, over-allocation profits are modified but only at the margin (they slightly increase with  $\overline{H}_2$ , and slightly decrease with  $\overline{H}_2$ ).<sup>39</sup> 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<sup>40</sup>. Our figures would then

<sup>39</sup> 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 over-allocation 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 over-allocation profits are at least 5MEUAs.

<sup>40</sup>  $\delta A_2^T - \delta E_2 = \delta A_2^T - \delta A_2^{OBA} + \delta A_2^{OBA} - \delta E_2$ .  $\delta A_2^{OBA} - \delta E_2$  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 large if it does cross (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

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 to relaxing  $H_1$  and  $H_2$ . The more 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).

### 3.5. Chapter conclusions and policy implications

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 over-allocation profits in case of downfall in the 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 over-allocation profits, it also created operational distortions which lead to outcomes inconsistent with the low carbon transition of EU energy intensive industries. The reduction in over-allocation 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 over-allocation 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 (Arjalies 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. In short, distortions generated

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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).

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.<sup>41</sup> Had we observed growth, the threshold rule may have been less relevant. Anecdotal evidence<sup>42</sup> suggests that instead, the reserve for new entrants may have been a more important source of distortions (there would be 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 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 paper suggests that this option would outperform both *ex ante* allocation with and without thresholds in terms of reducing distortions and over allocation 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 on-going discussions on how to circumvent these issues. For example

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<sup>41</sup> 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.

<sup>42</sup> Ref. private conversation with industry representatives.



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.

## 3.6. Chapter Appendix

### 3.6.1. EUTL Data computations

#### 3.6.1.1. Determination of the Activity Level Correction Factor (ALCF<sub>2013</sub>) at the plant level

The key challenge is to correctly distinguish installations that are above or below thresholds (25% and 50% of  $q/\text{HAL}$ ), 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 (1)):

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

Where  $CSCF_{2013}$  is the 2013 Cross Sectoral Correction Factor (0.9427),  $I_B$  the clinker carbon intensity benchmark (766 kg CO<sub>2</sub> 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:

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

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<sub>2</sub> 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<sub>2</sub> per ton of clinker to 1300 kg CO<sub>2</sub> per ton of clinker (and for the very large majority of the plants from 780 to 950 kg CO<sub>2</sub> 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<sup>43</sup>), 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  $q/\text{HAL}$  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  $q/\text{HAL}$  in low demand countries. This bias could also be explained by the clinker carbon intensity improvement between HAL years and 2012.

<sup>43</sup> In our data there is actually a gap between 2.14 and 4.01 so no case of overlapping.

### 3.6.1.2. 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<sub>2</sub> 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<sup>44</sup> clinker carbon intensity in big countries corresponds to GNR data in 2008 (818, 831, 832, 797, 847, 858, 849 and 842 kg CO<sub>2</sub> 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 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.

### 3.6.2. 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):

$$Q_C = C_C + NE_C$$

$$Q_K = R \times Q_C + NE_K$$

These equations must be verified for each country every year (for real or counterfactual scenario).

In this paper for real data,  $Q_K$ ,  $NE_K$ ,  $NE_C$  and  $C_C$  are obtained through different sources (see Table X), and  $Q_C$  and  $R$  are re-computed (we have  $R = \frac{Q_K - NE_K}{C_C + NE_C}$ ).

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<sup>44</sup> 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 ( $\bar{Q}_{K,i,2008} = I_{i,HAL} \times E_{i,2008}$ ).

### 3.6.3. Counterfactual clinker production and net trade estimations

#### 3.6.3.1. Plant level clinker production estimation

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  $PlantActivity_{i,t} \approx \frac{E_{i,t}}{HEL_i}$  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:

$$\begin{aligned} \Delta \ln PlantActivity_{i,t} &= \alpha_0 + \beta_1 \Delta \ln CementConsum_{c \ni i,t} + \beta_2 \Delta \ln GDP_{c \ni i,t} + \gamma_1 \ln RelativeCo2intensity_i \\ &\quad + \gamma_2 \ln RelativePlantsize_i + \gamma_3 Coast_i + \varepsilon_{it} \end{aligned}$$

In order to accommodate the autoregressive nature of plant activity, we define all country-level variables (source of the data is in Table 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 plant-level variables: the relative average carbon intensity of a plant<sup>45</sup>; relative plant size<sup>46</sup>; and a dummy variables for coastal plants.<sup>47</sup> 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.

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<sup>45</sup> 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 ( $RelativeCo2intensity_i = \ln(I_{HAL,i} / \overline{I_{HAL,c \ni i}})$  where  $\overline{I_{HAL,c \ni i}}$  is the average carbon intensity of plants (in tons of CO<sub>2</sub> per ton of clinker) in the country where the plant  $i$  is located.

<sup>46</sup> 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 ( $\ln RelativePlantsize_i = \ln(HAL_i / \overline{HAL_{c \ni i}})$  where  $\overline{HAL_{c \ni i}}$  is the average historical activity level (in Mt of clinker) in the country where the plant  $i$  is located.

<sup>47</sup> The dummy  $Coast_{it}$  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.

**Table 12: Regression results of corrections at the plant level**

	(1)
Log Cement Consumption	0.819*** (7.23)
Log GDP	0.235 (1.31)
Log Relative Carbon Intensity	-0.333*** (3.05)
Log Relative Historical Activity Level	0.013 (1.10)
Coastal dummy	-0.037*** (2.90)
Constant	-0.003 (0.34)
Observations	737
Plant level fixed effects	No
R2	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 12 column (1) shows the results for the period 2008-2011 (post-crisis). Cement consumption has a statistically significant 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 semi-parametric approach (Powell 1994) by specifically modelling the multiplicative error. The counterfactual plant activity level is then not fixed but is a random variable:

$$\begin{aligned} \tilde{PlantActivity}_i^{CF-2012} &= PlantActivity_i^{2011} \times \exp(\hat{a}_0) \\ &\times \left( \frac{CementConsum_{cs,2012}}{CementConsum_{cs,2011}} \right)^{\hat{\beta}_1} \times \left( \frac{GDP_{cs,2012}}{GDP_{cs,2011}} \right)^{\hat{\beta}_2} \\ &\times RelativeCO_2Intensity_i^{\hat{\gamma}_1} \times RelativePlantSize_i^{\hat{\gamma}_2} \\ &\times \exp(\hat{\gamma}_3 Coast_i) \times \exp(\tilde{\epsilon}). \end{aligned}$$

**Extending the smearing estimate of Duan (1983), we first fit the distribution of  $\tilde{\epsilon}$  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  $\tilde{\epsilon}$  (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 Table 5 and 6.**

### 3.6.3.2. 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.<sup>48</sup> This enables us to essentially extrapolate historic net 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:

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<sup>48</sup> 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.

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

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

For clinker net exports, the coefficient on  $\lambda_1$  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  $\lambda_1$  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:

$$\Delta NE_{K,j}^{CF2012} = NE_{K,j}^{2011} + \lambda_1 \Delta CementConsum_{c,2012}$$

and counterfactual net exports of cement as:

$$\Delta NE_{C,j}^{CF2012} = NE_{C,j}^{2011} + \mu_1 \Delta CementConsum_{j,2012}$$

It should be noted that the cement consumption was remarkably low in 2012. Because of the consumption/export relationship established by the econometric model, clinker net exports would have risen anyway in 2012 compared to 2011 had the threshold rule not be implemented.

## Chapter 4

# Carbon price pass-through in carbon-intensive materials sectors: Evidence from cement and steel price data under the EU ETS

### 4.1. Introduction

Do industries that produce carbon-intensive materials pass-through carbon costs to consumers in their product prices? This question has become the subject of increasing debate in European climate policy circles, as the EU has begun a new phase of reform of its Emissions Trading Scheme (“EU ETS”) (European Commission, 2015).

The EU ETS covers a number of so-called energy-intensive, trade-exposed industries, such as steel, cement and chemical production<sup>49</sup>. To protect against the risk that these industries may offshore production in order to avoid regulation (so called “carbon leakage”, the EU ETS Directive allows for them to receive an ex ante free allocation of emissions allowances each year in line with their historical production levels multiplied by a best available technology benchmark (EC/2003/87). The granting of such free allowances has significant distributional implications and therefore the rules on which these allocations are determined are controversial (cf. Sandbag, 2015). However, it is generally accepted by most experts that some form of anti-leakage provision is necessary for highly carbon or energy intensive sectors, such as cement, steel and aluminium, so long as global regulation has not sufficiently converged (European Commission, 2014).

However, one criticism of current anti-leakage provisions in the EU ETS is that, while they may limit carbon leakage, they may also be inconsistent with the incentives needed for deep transformation (i.e. decarbonisation) of the sectors concerned. In particular, it is argued that the decarbonisation of sectors producing carbon intensive materials – such as cement, steel and aluminium – depends critically on whether firms pass-through carbon costs to consumers in their product prices. Recent studies of emissions reduction potentials for some large EITE sectors like cement and steel have demonstrated that, based on current technology, it is currently impossible to achieve sufficiently deep cuts in emissions without substantial improvements to the way society consumes and uses carbon intensive materials. In short, we will need much greater efficiency in use of these materials by consumers, greater substitution between high and low-carbon materials in appropriate applications, and stronger economic incentives for product innovation that reduces the carbon footprint of the materials used (Neuhoff et al, 2015a; Neuhoff et al, 2015b).

There is also substantial complexity of the range of products, of end-use applications and of potential substitutes along the value chain in carbon intensive material sectors. This complexity makes it seem unlikely that appropriate levels of product substitution, material efficiency, and innovation support, could be mandated by a centralized regulator, e.g. via direct regulation and subsidy support. In short, a robust carbon price that is passed along the value chain in energy-intensive materials sectors seems

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<sup>49</sup> e.g. steel production, cement production, oil refining and petro-chemical production, as well as others such as pulp and paper, glass, ceramic and brick, aluminium, etc. For further details see [http://ec.europa.eu/clima/policies/ets/allowances/industrial/index\\_en.htm](http://ec.europa.eu/clima/policies/ets/allowances/industrial/index_en.htm)



necessary. This in turn raises the question of whether carbon prices are indeed passed on in product prices by EITE industries.

The policy logic followed by the European Union in granting free allocations to EITE sectors is essentially the following: sectors exposed to competition in international markets may struggle to pass on carbon prices and remain competitive. Thus, they may be granted free allocations, which allow them to avoid passing on their cost of carbon in their product prices. In response to this reasoning, some economists have argued that, in theory, ex ante free allocation should not necessarily prevent carbon prices being passed on to consumers (e.g. Bovenberg and Goulder, 2001; Burtraw, et al., 2002; Reinaud, 2003; Smale et al 2006). They note that free allocation in the EU ETS is granted “ex ante”, essentially as a lump sum payment of allowances at the beginning of each trading period. This means that any free allowances not used by firms for compliance could be sold on the carbon market at the prevailing carbon price. Thus, free allowances, when allocated ex ante, add an “opportunity cost” to each marginal decision to emit a tonne of CO<sub>2</sub>. In theory, a rational, profit-maximizing firm should treat opportunity costs like “real” costs and thus pass them on in their product prices.

However, there are several reasons why the “textbook” theory might not fully reflect the complexity of real-world decision-making. In practice, firm’s competitive strategies may not be fully captured by a static model that makes no allowance for market entry. In particular, EU firms may fear that a sustainable rise in prices above the international market equilibrium may create market entry that, in the long run, could be more damaging to their business models.

Further, firms may be conscious of the regulatory risks related to passing on the opportunity cost of free allowances. For example, in the EU ETS free allocation rules are redefined at frequent intervals (roughly every five years). Under the procedures established by the commission, the ability to pass through carbon costs is one of the key factors that is considered in defining who is eligible for free allocation. Firms may therefore fear that if they are observed passing on costs, they risk losing their entire free allocations in future years. Note that in the cement sector, this risk may be exacerbated by concerns about oversight from competition authorities. The cement sector in Europe is generally closely watched by competition authorities due for collusive practices and thus, as one executive explained, they would be loath to attract unwanted attention by a surge in profits resulting from the passing on of implicit costs of freely allocated emissions allowances.

Finally, it is also possible that firms simply do not have the power to sustainably pass on carbon costs in specific markets. For instance, many commodity metals (such as copper, steel or aluminium) are traded in very liquid international markets where price differences can quickly lead to shifts in competitiveness (e.g. London Metals Exchange). In the EU, given its low share of global markets, may therefore be unlikely to see local companies set prices that differ fundamentally from those set in the global market place.

Despite the importance of the price-pass-through question, there remains relatively little reliable empirical work on the subject. This paper therefore investigates this question empirically for the two largest such sectors in the EU ETS, namely cement and crude steel production. The analysis uses monthly price data in the EU markets for grey Portland cement and cold rolled coil (a commodity steel product). The results cast significant doubt on the hypothesis of price pass-through. If accurate, the findings suggest that the EU’s current approach to protecting ETS firms from carbon leakage is incompatible with the decarbonisation of carbon-intensive material sectors.

The rest of the paper proceeds as follows. Section 4.2 presents the most relevant empirical literature on the topic, provides a critical analysis of this literature, and situates this paper within it. Section 4.3

presents the econometric methodology and data. Section 4.4 presents the results of the main econometric specifications. Section 4.5 concludes.

## 4.2. The empirical literature

A number of papers have been published establishing that a significant share of EU ETS carbon prices are passed on into *electricity* prices (e.g. Sijm et al, 2006a, 2006b, 2008; Zachmann and Hirschhausen, 2008; Solier et al, 2011). These results are generally highly convincing, not only because of the high quality of the data used (daily, digitally recorded market exchange data and large sample sizes), but also because of the consistency of the results across different studies.

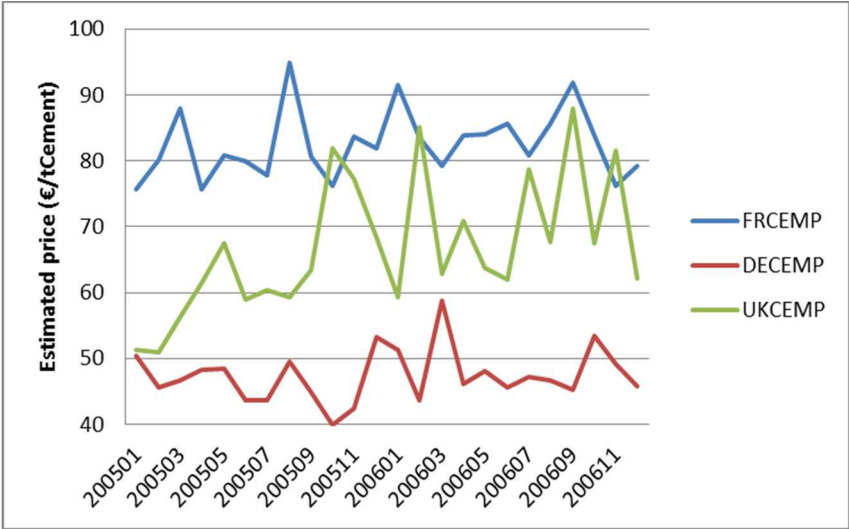
However, evidence of carbon price pass-through in electricity prices does not necessarily mean that the same phenomena can be assumed for EITE sectors. For one thing, many electricity companies were not fully allocated all of their allowances under the ETS in Phases 1 and 2, which means that the line between passing through an opportunity cost vs passing on a “real” cost is blurry in this literature. Second, European electricity producers do not face a threat of rapid foreign entry into their markets or loss of market share if they pass on opportunity costs, since electricity is to a large extent not subject to non-European competition.

To this authors’ knowledge, only a small handful of papers have empirically analysed the evidence of carbon price pass-through in energy-intensive trade exposed industrial sectors of the EU ETS. This first such paper is Walker (2006), which offered a first econometric analysis of the correlation between cement prices and carbon and other input prices in a number of EU Member States over the 15 months of the EU ETS. Since market cement prices are not publicly available, his analysis made use of the implied cement prices in Eurostat trade data, which reports both the value and volumes of cement imported by EU countries from other EU countries. The results suggested that cement prices in major EU economies such as France, Germany, Spain, Italy, and the UK, may pass through somewhere between 10 and 35% of CO<sub>2</sub> costs into final product prices.

However, serious questions can be raised about the robustness of these results. Firstly, the author of this paper noted that this result could be sensitive to the small data sample, which included only 15 (monthly) data points in which carbon prices were present in the sample.

Secondly, the cement price data were of questionable quality. The authors did not have access to explicit cement price data. Thus cement prices in the sample were derived implicitly based on data on cross-border trade values and volumes of traded cement. These data are, in our view, extremely unreliable as a guide to actual cement market prices. Indeed, in an attempt to replicate the results of this study, we re-derived these implicit cement price data for three EU member states, namely France, Germany, and the UK. As shown in Figure 4.1, these data are extremely volatile on a monthly basis. This raises significant doubts about the quality of these data as representative of actual cement market prices, and in particular about their usefulness for econometric analysis using small sample sizes.

**Figure 4.1 Implied “cement prices” (€/t) of imports into FR, DE, and UK (2005-2006)**



Source: author, based on Eurostat data.

Furthermore, discussion with EU cement industry experts revealed additional reasons to be sceptical of such data – namely that many trade flows have values that are not reported at market prices since they are not driven by market transactions, but rather by intra-firm transactions. Thus it is likely that such data muddle up market sale prices and intra-firm transfer prices, which can differ substantially.

Alexeeva-Talebi (2010) performed an econometric analysis of price pass-through the petroleum refining sector, using a Vector Error Correction Model (VECM). This study used weekly carbon price data and Euro95 unleaded petroleum price data in 14 Member States during the period of September 2005 to March 2007. It concluded that, on average, a 1% increase in the carbon price was associated with a 0.01 to 0.09% rise in European petroleum prices, which equates to long run pass-through rates of between 25 and 113%.

Again, however, a number of caveats need to be placed on this finding. Firstly, and most importantly, the paper’s stated conclusions appeared not to be completely supported by the reported regression results. For example, the reported results indicated that the estimated carbon price pass-through coefficient was statistically significantly different from zero (at conventional significance levels) in only 3 out of the 14 Member States. A number of coefficients were found to be the right mathematical sign and of plausible economic magnitudes. However the lack of statistical significance suggests that conclusions are at best tentative. Secondly, the estimation methodology ignored the potential role of demand side variables in influencing prices of both unleaded petroleum and carbon allowances. This creates a risk that both unleaded petroleum and carbon prices could have been correlated with broader economic demand factors. For these reasons, the results cannot be considered definitive.

Perhaps the most recent and comprehensive paper to analyse the question of carbon price pass-through in traded sectors of the EU ETS is De Bruyn et al (2011). They also used a VECM approach. They regressed weekly or monthly EU market prices of specific steel, chemical, and refined petroleum products on their US counterparts and on the price of European Union Emissions Allowances (“EUAs”). Their results suggested that European prices of cold-rolled steel, hot-rolled steel, gasoil and diesel consistently include a pass-through of approximately 100% of EUA costs, sometimes with a lag of

several weeks. They also suggested that EU prices of polystyrene, polyethylene and polyvinylchloride exhibit a pass-through of approximately 30% of EUA costs.

Once again, however, difficulties with the analytical methodology suggest that these results need to be considered with caution. A significant limitation of de Bruyn et al (2011) is that it didn't effectively control for the effects of possible differences in energy input prices and market demand between the EU and US. This unfortunately casts doubt on the reliability of the estimates because of possible omitted variable bias. For example, the EU price determinant literature shows EU primary energy prices are correlated with the prices of EUAs (Ellerman et al, 2010). Similarly, EU energy prices are likely to be correlated with prices of energy-intensive goods in the EU (via production costs). Thus it is possible that the estimates of the effects of the EUA prices on final product prices for energy-intensive goods may actually reflect the impact of EU *energy price* changes on final product prices, rather than the impact of *EUA* price changes. A similar argument can also be made regarding EU market demand, which, as is not explicitly controlled for.

Finally, de Bruyn et al (2011) do not explicitly take into account the distinction between price pass-through of CO<sub>2</sub> via electricity purchases, as compared to the price pass-through related to direct emissions. Free allowances were allocated to energy-intensive industries in the EU ETS in Phases 1 and 2 for *direct* (onsite) emissions but not for indirect emissions (i.e. emissions higher up the value chain stemming from the rise in electricity costs that occurred when the carbon price was introduced). This means that, even if one finds evidence that a part of carbon prices are being passed on in product prices, one needs to control explicitly for electricity prices as a separate variable in the regression analysis to separate out the relevant effects. This wasn't done in this analysis.

In short, the above 3 papers each have significant questions that can be raised about the underlying methodologies and this casts important doubts on their conclusions.

This doubt is also added to by a closely related strand of the carbon market literature, specifically by a set of empirical papers looking at *firm-level impacts* of the EU ETS on EITE firms. Specifically, high levels of free allocations that EITE firms received in the EU ETS imply that firm profits should have increased if firms were passing on their carbon costs into product prices, after controlling for other factors. However, the literature does not identify such a result. For example, Anger and Oberndorfer (2008) find no statistically significant effect of the level of free allocation on the revenues of German non-electricity firms in 2005 and 2006 compared to 2004. This result was also found in a working paper by Abrell et al (2011)

There is also range of studies which look at price pass-through of other – i.e. *non-carbon price* – cost factors, such as exchange rates and other input cost changes in manufacturer output prices. For example, Alexeeva-Talebi (2010b) provides a thorough and convincing analysis of price pass-through in energy-intensive sectors in the German case, and Oberndorfer et al. (2010) which did the same in the UK. Interestingly, this empirical literature tends to conclude that strategic considerations about maintaining market share are a key factor in price pass-through rates in manufacturing sectors (e.g. Knetter 1993, Clostermann 1996, Goldberg and Knetter 1997, Stahn 2006 and Gaulier et al. 2008). This may in turn help to explain why full carbon price pass-through of opportunity costs might not occur in the presence of free allocation, as found by Walker (2006) and as suggested by the results Anger and Oberndorfer (2008).

In summary, the above survey suggests that the empirical literature is at best inconclusive on whether EU ETS firms in traded sectors pass on carbon costs to consumers in their product prices. The present paper therefore revisits the evidence of price pass-through in two sectors where price pass-through

effects should be expected to be very visible if they do indeed exist – namely the steel and the cement sector. This is done in an attempt to respond to some of the uncertainties which have been identified in relation to the above studies. In particular, we seek to exploit the passage of time to use larger data samples, to use more reliable data sets on product prices, and to control for key omitted variables in the relevant markets, such as demand, electricity prices, and fossil fuel prices. The following chapter presents these data and the econometric methodology.

### **4.3. Methodology**

#### **4.3.1. Price fundamentals in the cement and steel sectors in Europe**

##### **4.3.1.1. Cement price fundamentals**

The EU cement sector is mainly characterized by local production for local markets. This is due to the low value/tonne of product of cement compared to other products, which make it expensive to transport, especially by road. Thus, competition from imports is generally low (Neuhoff et al, 2015), but can be more intense in specific coastal locations where dedicated import infrastructure exists (Branger et al, 2015). The localization of production means that prices can vary significantly across regions of Europe (Walker, 2008). For the econometric analysis, this implies that local rather than global factors are more likely to influence average prices in a specific country or region in Europe.

There is also high degree of market concentration in cement markets. Production in the major national markets in Europe is usually dominated by between two and four producers (Boyer and Ponsard, 2013). The sector could appropriately be described as exhibiting many characteristics of oligopolistic competition. Basic industrial organization theory suggests that pass-through rates of costs will tend to be less than 1 for 1 in firms under oligopolistic competition, since firms will not necessarily set price equal to marginal costs (Frank, 2004).

The cement industry in Europe has experienced periods of stable collusion followed by price wars in its recent history, such as in the German cement market in 2002-2003. The UK Competition Commission also recently found that price competition is also not as intensive as it may be in the UK cement sector due to the intensive knowledge of each participant of others' pricing behavior<sup>50</sup>. It argued that this creates favourable conditions for "tit-for-tat" pricing practices that can undermine competition among market rivals on price. There is a body of empirical evidence which suggests that, where pricing is coordinated or collusive, price volatility is reduced, as greater price volatility can make enforcement difficult and can lead to the breakdown of coordination (Cf. Hüschelrath and Veith, 2011).

One possible consequence of collusive price behavior is that it can lead to asymmetric responses of prices to input costs or other exogenous shocks. Specifically, prices may be less volatile in response to input price falls, while input price rises can be more easily passed on without fear of retaliation. This phenomenon, which has long been identified in other markets, such as retail gasoline prices, is known

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<sup>50</sup> For example, the UK Competition Commission provisionally found in 2013 that "coordination between the three major cement producers in the [UK] cement market is likely to be resulting in higher prices for all cement users". It noted that "the CC's finding does not relate to explicit collusion between these producers. Rather, as the cement market is highly concentrated with only four GB producers (Hope Construction Materials (HCM) being a new entrant), who have an unusually high level of understanding of each other's businesses—this has created conditions which allow three of them to coordinate their behaviour, thereby softening competition and resulting in higher prices for consumers". <http://www.competition-commission.org.uk/media-centre/latest-news/2013/may/cc-looks-to-break-open-cement-market>

as “rockets and feathers”. We therefore test for this hypothesis using a separate specification in the econometric analysis.

Variable production costs in the cement sector in the EU are mainly driven by a few key factors, including: the cost of primary fuels, particularly gas and oil, which is used to heat cement kilns; the cost of electricity, which is used to drive grinding, crushing and mixing activities and other equipment at the plant; transport costs; and the cost of labour. There are good reasons to think these variables are correlated with CO<sub>2</sub> prices themselves. We therefore controlled for these key variables explicitly in the model we estimated. Other factors, such as the cost of limestone, gypsum, aluminium, calcium carbonate, and aggregates, such as fly ash, pozzolana, blast furnace slag also play a role. However, in CEM I – which is basic Portland cement and the major cement traded in Europe – these costs are less important to total costs. There is also less evidence to think that they may be correlated with CO<sub>2</sub> prices.

Finally, it should be noted that cement production is a highly CO<sub>2</sub> intensive product and thus at high CO<sub>2</sub> prices, one should expect the evidence of CO<sub>2</sub> price pass-through to be very clear. Each tonne of cement production in the EU typically results in between 0.6 and 0.8 tCO<sub>2</sub> emissions (GNR, 2016). Average cement prices in Europe per tonne of cement vary in the range of 50 to 100€/t in western Europe in recent years. Meanwhile, EU ETS carbon prices have moved in the range of 0 to 30 €/tCO<sub>2</sub> during the past 10 years. This implies that, at their peak, CO<sub>2</sub> costs could have been in the order of 18% to 48% of cement prices during this period. *Thus, one would expect the impact of even a partial pass-through of CO<sub>2</sub> opportunity costs in cement prices to have been reasonably clear in the data if indeed it did occur.*

#### **4.3.1.2. Steel price fundamentals**

A wide range of steel products are traded in the European steel market. For our purposes in this paper, these products can be categorized into two groups: “commodities” and “specialities”. Commodity steel products like basic flat products (hot and cold rolled coil, plate, strip, etc.) and basic long products (bars, beams, tubes and wire) tend to travel easily across borders and are typically traded in liquid international commodity markets, such as on the London Metals Exchange.

Commodity steel products therefore tend to show a high degree of price correlation between key international markets, although transport costs can still explain some cost divergences between the key regional markets, as highlighted by data series collected by international market data providers ([www.Agmetalmminer.com](http://www.Agmetalmminer.com)). The main regional markets in the steel sector are Europe, US, China and Japan, although Middle Eastern and other Asian markets are becoming more relevant.

An implication for the analysis is that there are likely to be more or less stable long run relationships between commodity steel prices in the EU and other key regional markets. In econometric terms, these long run relationships could potentially be so-called “co-integrating” relationships between EU steel prices and steel prices in other major regional markets. Consequently, there would also be a vector error correction model form of these cointegrating relationships that would describe the short run adjustments of the prices in these markets back to their long run equilibrium in the presence of exogenous shocks (Engel and Granger, 1987).

In the context of our analysis, if EU firms pass on the opportunity cost of CO<sub>2</sub> prices in the EU ETS in their own prices, then the CO<sub>2</sub> price movements would in fact represent a specific example of such exogenous shocks on the long run relationships between prices in EU and other steel markets. This has important implications for the econometric specification to be adopted for estimating carbon price pass-through behavior in the EU steel sector.

Variable cost drivers of commodity steel prices are a little bit more complicated for steel production than for cement, as there is more than one route to produce equivalent quality commodities of steel and these can vary a little bit depending on the type of steel making process used. Two main production techniques account for over 94% of global steel production (OECD, 2013). These are the Blast Furnace/Blast Oxygen Furnace (BF/BOF) route and the Electric Arc Furnace (EAF) route. The former accounts for over 70% of global steel production, and is the main driver of price fundamentals in global markets. Using this route, the majority of variable cost is driven by the cost of iron ore and coking coal (typically over 50% of total variables costs (WikiInvest, n.d.)). The remainder of costs being composed of smaller cost components, such as steel scrap, electricity and labour, fluxes and industrial gases. Using the EAF route, the main costs are the cost of steel scrap, which substitutes for iron ore, while electricity consumption is more important and substitutes for coking coal as the primary energy input.

Some of these cost variables influencing EU steel production costs are potentially correlated with CO2 prices, while others are not. Specifically, electricity prices, oil prices and coal prices have been shown to be correlated with CO2 price fundamentals (cf. Chèze, 2009). These are therefore controlled for explicitly in the model described in the next section.

## 4.4. Econometric specifications

### 4.4.1. Cement

#### 4.4.1.1. Main Model specification

Based on the above discussion of cement price fundamentals, we developed the following reduced-form equation to describe short run (monthly) cement price formation fundamentals:

$$(1) \quad PCEM_{it} = \alpha.PCO2_t + \Sigma_1.POIL_{it} + \Sigma_2.PELEC_{it} + \Phi.PLABOUR_{it} + \beta.CONSTR_{it} + \delta_4.EURX_t + \Omega.PMETALS_{it} + \Pi.MONTH_t + FE_i + u_{it}$$

Where,

- $PCEM_{it}$  is the average monthly cement price within country  $i$  in month  $t$ .
- $PCO2_t$  is the average monthly price of EU ETS carbon emissions allowances to emit one tonne of CO2.
- $POIL_{it}$  is the average monthly price per barrel of Brent crude oil (an EU benchmark).
- $PGAS_{it}$  is the average monthly price/mmbtu of natural gas in Germany (a western Europe benchmark)
- $PELEC_{it}$  is the average price of electricity delivered to for large industrial consumers in country  $i$  at month  $t$ .
- $PLABOUR_{it}$  is a monthly labour cost index for industry in country  $i$ , based on effective exchange rate indices.
- $CONSTR_{it}$  is a monthly measure of national construction activity index for country  $i$

- $EURX_t$  is a measure of the EU nominal exchange rate compared to a trade-weighted basket of 27 major EU trading partners.
- $PMETALS_{it}$  is a measure of the prices of EU metals, based on a global metals price index.
- $MONTH_t$  is a vector of dummy variables for different months of the year capturing seasonality effects
- $FE_i$  is a term representing country fixed effects.
- $u_{it}$  is an error term capturing unobserved factors affecting monthly cement prices in these three countries.

This list therefore represents a systematic breakdown of the main exogenous supply and demand side variables influencing cement prices and which could potentially be correlated with CO2 prices. On the supply side, it includes the main cost factors that affect cement prices (energy prices, labour costs<sup>51</sup>), while on the demand side it includes the main exogenous factors one would theoretically expect to influence prices (construction demand for cement, exchange rates and thus trade competition, the price of close substitutes (metals), and seasonal effects. Furthermore, the fixed effects term should capture any time invariant heterogeneity in cement prices across the 3 countries in the sample, namely Germany, UK and France, for instance due to different levels of internal or external competition, regulatory conditions, etc.

Note that these factors are all *a priori* exogenous in the sense that there is no obvious theoretical reason to think that the explanatory variables themselves should be determined by cement prices. Against this, one could attempt to argue that cement demand (i.e. here measured by construction activity) could be determined in part by cement prices. However, we would dispute that this has a material impact on our estimation. Firstly, in general cement is a minor part of total construction costs for the most common usages (e.g. building construction, cf. NAHB, 2016). Second, during the period of our analysis, i.e. 2005-2015, fluctuations in the demand for cement in Europe has largely been driven by macro-economic factors, such as the housing bubble and bust in 2008/09 and the subsequent economic crisis in the Eurozone. We are therefore confident in assuming demand to be an exogenous factor acting on prices during the sample period.

The parameter estimate of interest in the above model is therefore  $\alpha$ , as this variable should represent the rate of pass-through of CO2 prices into cement prices.

In addition to the above model, we also developed two further variations. Firstly, we also estimated a model based on the possibility of a “rockets and feathers” effect on cement prices. In this model, we differenced the data and split each of the key variables into two separate variables. Thus for instance,  $\alpha.PCO2_t$  was split into two separate terms, where  $\alpha^+.PCO2^+_t$  captured positive price impulses only, while  $\alpha^-.PCO^-_t$  captured negative price impulses only in order to allow for differences in the two effects.

Secondly, to allow for potential differences in the level of cost pass-through across the three examined countries we estimated the each of these models for each of the three countries individually.

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<sup>51</sup> Note that while there is no empirical evidence that labour costs are correlated with CO2 prices in the CO2 price fundamentals literature, there is evidence that CO2 prices are correlated with GDP growth and industrial output (Chèze et al, 2009), and these in turn can be correlated with labour cost growth.



#### 4.4.1.2. Data

Monthly data on cement prices for the period 2005 to 2015 inclusive were collected for the three largest cement producing countries in Europe, namely France, UK, and Germany. This period corresponds to the period of the introduction of the EU ETS (which began in January 2005). Data for France came from INSEE's Industrial Product Price Database, which monitors cement prices as an index (rather than as absolute prices). This index was subsequently converted into prices using information on cement prices in specific months that came from discussion with industry experts. The same process was done for the UK, where monthly construction cost data (including cement prices) have been monitored by the Department of Business Innovation and Skills' Construction and Material Cost Database since 2005. Data for German cement prices were computed in the same way and came from the German Federal Statistical Office.

CO2 allowance price data is based on December delivery European Union Allowances (EUAs) as reported by London's ICE exchange (the most liquid CO2 allowance product in the carbon market). Brent crude oil prices, German natural gas prices, and metals prices came from the IMF's Commodity Price Database. National industrial electricity prices for consumers of over 70MWh per year came from Eurostat's data on energy prices. Labour costs were proxied by a measure of labour cost inflation based on real effective exchange rates for all three countries from Eurostat. National construction indices came from Eurostat's Short term business statistics (STS) database. Nominal exchange rate data also came from Eurostat and was based on trade-weighted average exchange rates with 27 major trading partners.

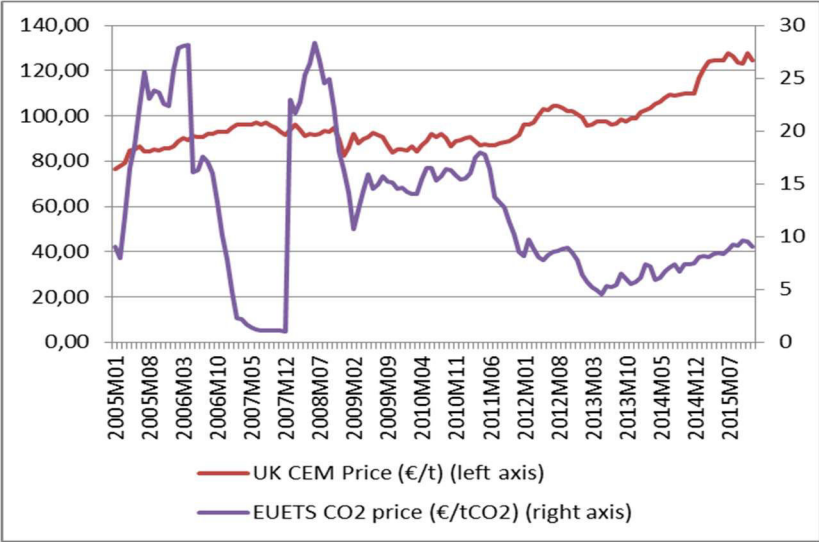
#### 4.4.1.3. Preliminary Data Analysis

Preliminary data analysis showed that the cement price data were non-stationary, as illustrated by the example of the UK below (cf. Figure 4.2

Figure ). Johansen cointegrating rank tests revealed little evidence of a co-integrating relationship between cement prices, CO2 prices and our main explanatory variables. We therefore obtained first differences of the variables, which provided stationary data. Breusch-Pagan tests revealed little evidence of heteroscedasticity. However, some serial correlation was observed. Further testing revealed that an AR(1) term should be included in the regression equation. This proved sufficient to generate white noise errors. Preliminary testing also revealed evidence of lagged impacts of explanatory variables on the

regressand. We therefore included specifications with up to 2 lags in the equation, with the lag-length chosen based on the Schwarz Information Criterion.

**Figure 4.2 Cement prices and CO2 prices in the UK (2005-2015)**

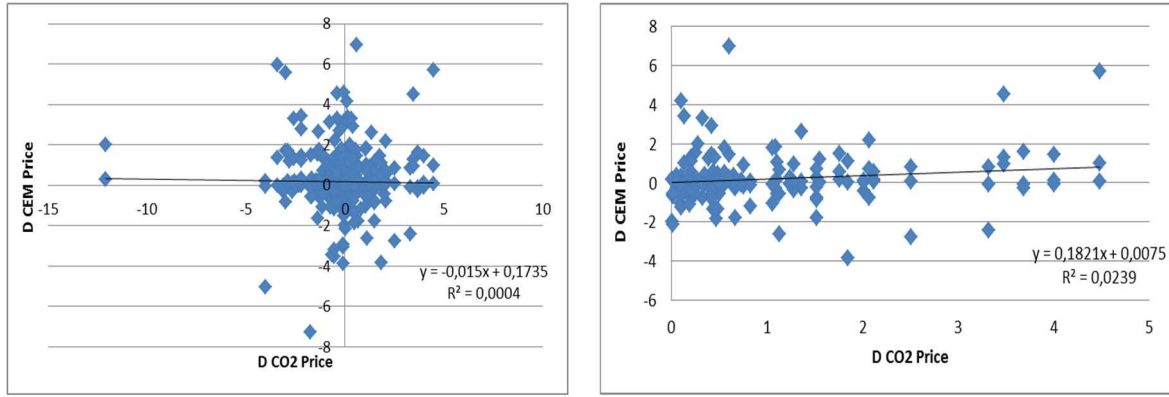


Preliminary analysis of the correlation between monthly changes in EUA prices and the cement prices showed relatively little evidence of a clear and positive correlation for the sample as a whole. Evidence of “rockets and feathers” was also limited (see 4.3).

**Figure 4.3 Plot of monthly changes in CO2 prices and cement prices**

**4.3.1 Full sample**

**4.3.2 Positive CO2 price changes only**



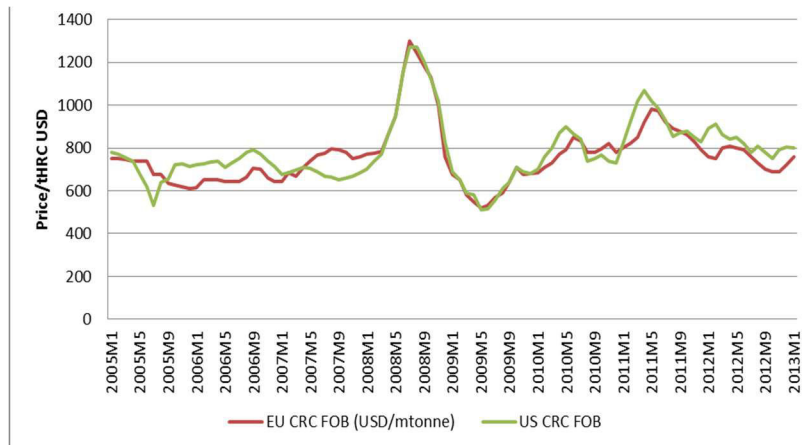
## 4.4.2. Steel

### 4.4.2.1. Main model specification

Based on the discussion in Section 2 on steel price fundamentals, we pursued an analysis based on the hypothesis of co-integrating relationships between steel prices in the EU and international markets.

Firstly, as a proxy for the international price of steel we used US steel prices for cold rolled coil (CRC). Based on preliminary observations of the data we posited the existence of a co-integrating relationship between Western-EU and international (US) CRC prices (Figure 4.4).

**Figure 4.4 US and Western EU prices of hot-rolled coil**



We posited that there was a cointegrating relationship between EU and US steel prices of the following form:

$$(2) \quad P_t^{EU} = a_1 + b_1 \cdot P_t^{US} + u_{1t}$$

$$(3) \quad P_t^{US} = a_2 + b_2 \cdot P_t^{EU} + u_{2t}$$

Where  $P_t^{EU}$  is the price of cold rolled coil in the (western) EU in month  $t$  and  $P_t^{US}$  is the same in the US in month  $t$ .

Second, based on this relationship, we hypothesized that a vector error correction model exists of the following form:

$$(4) \quad \Delta P_t^{EU} = \Delta P_t^{US} + \phi(P_t^{EU} - b_1 \cdot \Delta P_t^{US} - a_1) + \delta \cdot \Delta P_t^{CO_2} + \sum_{i=1}^n \sum_{j=1}^m \lambda_{t-j,i} \cdot Z_{t-j,i} + \varepsilon_t$$

Where the observed monthly change in the price of cold rolled coil in the EU is assumed to be a function of contemporaneous changes in US prices, an error correction term,  $\phi(P_t^{EU} - b_1 \cdot \Delta P_t^{US} - a_1)$ , that adjusts for any short run disequilibrium from the long run co-equilibrium relationship between US and EU prices, and a short run exogenous shock to EU prices stemming from any change in the level of European CO<sub>2</sub> prices,  $\delta \cdot \Delta P_t^{CO_2}$ , where  $\delta$ , represents the immediate short run pass-through rate of CO<sub>2</sub> prices into EU prices. This represents the main variable of interest for this analysis.

Finally, we include a vector of lagged control variables. These include controls on any variables that could potentially be correlated with CO<sub>2</sub> prices, including: the price of Brent Crude oil, the average price of industrial electricity for large consumers in Western Europe, the FOB price of Australian coal, construction demand in the Eurozone, labour costs, Eurozone exchange rates. These variables came from the same sources as described above for the case of the cement data, although the scope of the data was altered to reflect the entire region of Western Europe (the EU12), rather than just 3 individual countries. Prices for cold rolled coil in the US and Western Europe came from [www.MetalsBulletin.com](http://www.MetalsBulletin.com), a well-known steel market data provider.

#### *Preliminary Data Analysis*

A Johansen cointegrating rank test strongly indicated that there was one cointegrating equation between EU and US prices of cold rolled coil (cf. results reported in Annex).

Several specifications were estimated. The specifications of the cointegrating equation including the price of CO<sub>2</sub> in the long run relationship were *not* statistically significant at standard levels. Thus the CO<sub>2</sub> price was not included in the cointegrating relationship itself.

Data also revealed evidence of statistically significant lags of the regressand, suggesting an AR(1) process. Autogressive terms were therefore included to clean up serial correlation in the residuals. Using the Hannan-Quinn Information Criterion (HQIC) and Schwarz-Bayesian Information Criterion (SBIC), lags of just one month were deemed an optimal specification of the model. These were therefore included in the main specification that was estimated.

## **4.5. Results**

### **4.5.1. Cement results**

The results of the main regressions that were estimated using the cement price data are presented in Table 4.1 and 4.2 below. Table 4.1 presents both the results of the main Fixed Effects specification and an additional Random Effects specification as a robustness check, as well as country specific results for individual country-level regressions. Table 4.2 presents the results of the “rockets and feathers” specification that was estimated as a robustness check on the main results.

**Table 4.1 Regression results (cement)**

Parameter	Panel (FE)	Panel (RE)	UK (OLS)	FR (OLS)	DE (OLS)
PCO2 (S.E.)	.016 (.023)	.013 (.024)	-.064 (.061)	.057 (.040)	.009 (.035)
PCOAL	.010 (.008)				
POIL	0.018*** (.008)	.021*** (.008)	.060*** (.020)	-.003 (.007)	0.009 (.006)
PELEC	.116*** (.018)	.115*** (.018)	.092*** (.030)	-.139 (.085)	-.011 (.021)
PLABOUR	.600*** (.079)	.615*** (.078)	.872*** (.404)	.363*** (.171)	0.011 (0.061)
EURX	-8.56*** (1.97)	-8.46*** (1.98)	-30.9*** (5.36)	1.51 (1.77)	-.320 (1.32)
METALS	.006 (.008)		.027* (.189)	-.007 (.007)	-0.002 (.004)
CONSTR	.010 (.042)	.021 (.041)	.151 (.148)	.008 (.030)	.027 (.025)
JAN	.104 (.503)		-.871 (1.20)	2.17*** (.54)	-0.038 (1.04)
AUG	.206 (.220)		.318 (.512)	.450*** (.208)	.055 (.057)
L.PELEC	.032 (.031)	.111*** (.018)	-.042 (.021)	-.076 (.067)	.023 (.070)
L.PLABOUR	.194*** (.080)	.211*** (.078)	.492*** (.182)	-.056 (.087)	0.421 (.839)
L.PCO2	.005 (.026)	.005 (.023)	.067 (.061)	-.013 (.013)	-.006 (.018)
R-sq (within)	.289	.308	.44	.69	.032
N	393	393	131	131	131
P>F(6,384)	.000	.000	.000	.000	.064

\*\*\*Statistically significant at 99%, \*\*Statistically significant at 95%, \*Statistically significant at 90%. Standard errors in parentheses.

The models for which the results are presented in Table 4.1 were strongly jointly statistically significant. The overall explanatory power is reasonable, given that the data are relatively high frequency (monthly) and pooled across 3 countries.

The panel data estimations tended to find that key cost inputs into cement production were statistically significant determinants of prices. Specially, the price of oil, electricity, and labour costs were all found to be the expected sign and statistically strongly significant, and to be of reasonable economic magnitudes. (For instance, a 1c/Kwh increase in the price of electricity was found to be associated with

approximately a 1€/t rise in the price of cement). One-month lags of the latter two variables were also found to be statistically significant in some cases.

On the demand side, the strength the euro exchange rate was also found to be statistically significant. This variable appears, however, to be picking up changes in the broader economic context and thus in demand for cement. Indeed it appears to have swamped other measures of industry demand, such as construction output index, which explains why this variable was not jointly significant. The relative price of metals and cement was not found to have any meaningful explanatory power in the various models. In the joint models, the seasonal dummy variables did not show much explanatory power either, although in the specific case of the country level data for France the months of January and August proved to be strongly statistically and economically significant.

Regarding the main variable of interest, *the price of carbon was generally found to be of a positive sign but to be statistically insignificant* at anything close to conventional levels. The only possible exception to this finding was the individual case of France, where the parameter estimate on the price of CO<sub>2</sub> was statistically significant at the 80% confidence level. However, even this is far from convincing as evidence of a strong impact on cement prices. It should be noted that the economic significance of the parameter estimates is also very small across all of the models. For example, in France, the parameter estimates on the price of CO<sub>2</sub> would imply that a 1€ increase in the price of carbon allowances was associated with a 5.7c/t increase in the price of cement. This would imply a carbon cost pass-through rate in the order of 3 to 4%, which seems economically implausible.

It should be noted also the country level results for Germany, the UK and France also show a high degree of heterogeneity in terms of which factors were found to be significant at explaining cement price drivers. This suggests that either the data are unreliable, or, what we think is more likely, that there are other important factors driving cement prices that are not being captured by our model(s).

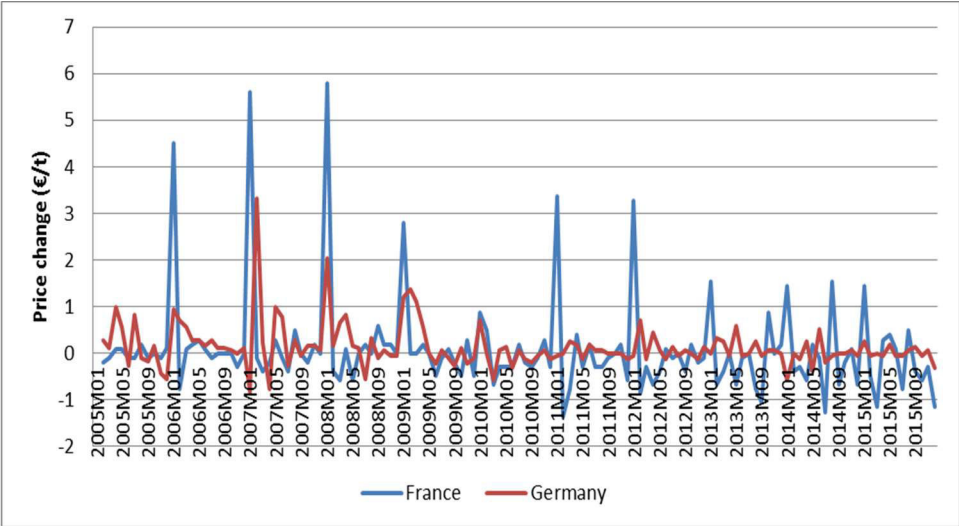
Closer inspection of the price data series for each of the individual countries showed a very different pattern in France and Germany compared to the UK. In the case of both Germany and France, pricing behavior appears to show a sudden rise at the beginning of each calendar year, followed by relatively stable prices during the year, while the UK price series appears to fluctuate more. What explains this marked difference across countries?

Two factors appear most plausible as explanations of these results. Firstly, as noted above, the main inputs into cement production, in terms of cost, are electricity, labour, and primary fuels. These inputs are often purchased on the basis of annual contracts that are set at the beginning of the year, or are linked to regulated prices or branch agreements that also follow a yearly cycle. This is particularly the case in France where the beginning of the year effect seems most significant, but also true to some extent of Germany and the UK. Moreover, UK energy prices have tended to be more volatile during the sample period. Thus, there has been lower intra-annual volatility of these prices in France and Germany compared to the UK.

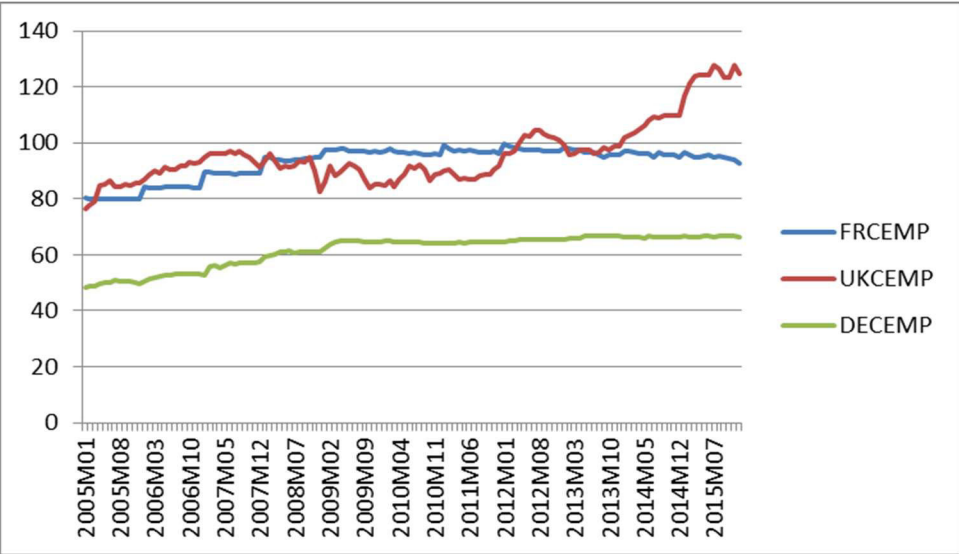
Moreover, the generally low level of intra-annual volatility in France and Germany could quite plausibly also reflect weaker competition (or indeed tacit collusion) in these markets. Aside from explicit collusive incidents, it seems likely that the generally non-competitive nature of the industry and high level of knowledge between firms on how they set prices, means that there is usually little pressure on prices to move in response to short-term changes in market conditions. The UK is to some extent an exception in this regard, due to a higher concentration of companies, and more volatile energy prices in recent years (especially for electricity).

Thus, abnormally low volatility in prices in these countries' markets, may make it hard for the estimation procedure to "identify" the impact of each of the relevant variables on the price of cement in Germany and France.

**Figure 4.5 Seasonal volatility in French and German cement prices (month-on-month changes)**



**Figure 4.6 Cement price data (UK, DE, FR)**



However, these data difficulties do not necessarily seem to undermine the conclusions of the evidence presented here. The data still quite clearly suggest that companies in the cement sector *do not* pass on the opportunity costs of carbon allowances. For instance, even setting aside the data for France and Germany above, the country-level data for the UK shows a lot more variability over time but a similar conclusion is reached.

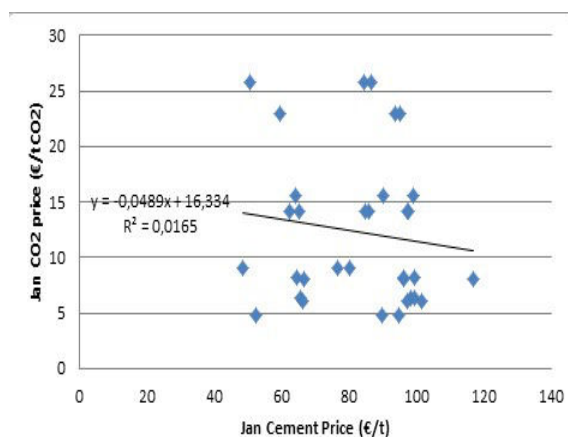
Further, even if one ignores short frequency movements in prices in Germany and France and focuses on the *levels* of prices, there is still relatively little evidence that annual changes in cement prices are



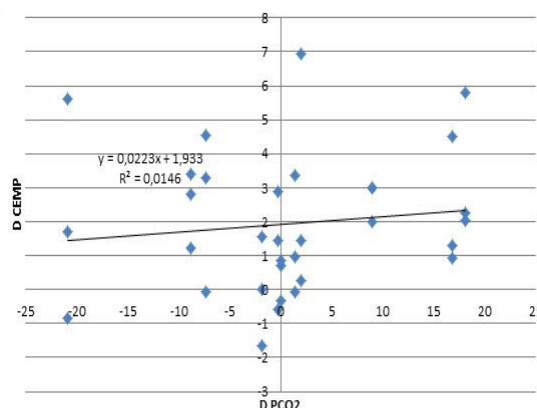
correlated with *the level* of CO2 prices. As noted above, preliminary estimates showed no evidence of co-integration between carbon and cement prices. Furthermore, even if one compares January cement price changes and CO2 price levels, one finds little correlation. **Erreur ! Source du renvoi introuvable.** 4.7 below compares the January price rises in cement with the *level* of CO2 prices in those months in France and Germany. On some of these occasions, CO2 prices were fairly high (around 25€/tCO2), yet we rarely see evidence of the +15 to +20€/t price hikes that should be visible if firms were passing through the majority of the cost of CO2 allowances. On the contrary, a linear correlation of the *levels* of cement prices and CO2 prices is negative. While annual changes shows a positive relation of just ~0.02 x carbon price (well below the 0.6-0.8 emissions factor of cement to CO2). In short: if cement firms are passing on CO2 costs in their prices, they appear to be hiding it pretty well.

**Figure 4.7 Are levels of cement prices correlated with the level of CO2 prices?**

**4.7.1 Cement price levels vs CO2 levels in January**



**4.7.2 Annual cement price change (yoy Jan) vs Annual CO2 price change (yoy Jan)**



**4.5.1.1. Rockets and feathers?**

The analysis of the “rockets and feathers” models was relatively inconclusive. This is mainly because the models suggests that there is little evidence that prices actually can be accurately modelled by such a model. The coefficient estimates tended to reject the rockets and feathers hypothesis by revealing a number of counter-intuitive and highly implausible results. For example, *negative* CO2 and oil price changes were found to be *positively* correlated with cement price rises, which is highly implausible and inconsistent with the underlying hypothesis of the model. Also, electricity price *falls* were *more strongly correlated* with cement price *falls* than electricity price rises were with cement price *rises*; and many coefficients were the wrong sign. Similar results were also found in the country specific regressions, although they are not reported here. This model also did not add any additional explanatory power to our simpler and more parsimonious model described above. Thus, it was rejected as having little explanatory power generally, and thus we treat the coefficient estimates – which are weakly significant for CO2 prices – with extreme caution.

**Table 4.2 Regression Results ("Rockets and Feathers" estimation, cement)**

	<b>Model (1)</b>	<b>Model (2)</b>
PCO2+	.064	.065**
(S.E.)	(.049)	(.030)
PCO2-	-.101***	-.091*
	(.025)	(.047)
POIL+	-.004	-.001
	(.003)	(.017)
POIL-	-.037*	.030**
	(.028)	(.015)
PELEC+	.077***	.067***
	(.012)	(.022)
PELEC-	.203***	.242***
	(.040)	(.043)
PLABOUR	.580***	.528***
	(.218)	(.177)
EURX	-9.11	-9.21***
	(10.69)	(2.16)
METALS	.003	.000
	(.009)	(.002)
L.PCO2+		.010
		(.030)
L.PCO2-		.000
		(.046)
L.PELEC+		-.041
		(.044)
L.PELEC-		-.015
		(.022)
L.PLABOUR		.166**
		(.079)
L.POIL+		-.002
		(.016)
L.POIL-		-.003
		(.015)
N	393	390
R-sq	.338	.358
P(F)>0	.000	.000

\*\*\*Statistically significant at 99%, \*\*Statistically significant at 95%, \*Statistically significant at 90%. Standard errors in parentheses.

## **4.5.2. Steel results**

### **4.5.2.1. Regression results and interpretation**

The results of the analysis of (western) EU cold-rolled coil steel prices is presented in Table 4.3. The Table shows the results of the co-integrating equation, three different vector error correction models that were estimated on the basis of this equation, as well as two more experimental models that were estimated in addition as “robustness checks”. The latter included a vector auto-regression model (VECM) of monthly changes in European CRC prices, and an OLS regression on EU steel prices based on a set of possible price determinants<sup>52</sup>.

The results indicated a single co-integrating relationship between EU and US CRC prices (the parameter estimates of which are described in column 2 of the table). The HQIC and SBIC suggested a VECM with one lag was the optimal parameterization of the model (cf. Annex for details).

The VECM results indicated that the adjustment parameter on the lagged errors was strongly statistically significant. The parameter estimates were in the order of -0.126 to -0.19, suggesting that the re-equilibration of the EU-US CRC price relationship was relatively slow. The results also confirmed a strong short term relationship between EU and US prices. They showed that, using monthly data, EU CRC prices tended to move contemporaneously with US CRC prices to a large extent, with between 50 and 70% of any price move in the US being reflected in EU prices, when evaluated at one month intervals. There was also strong evidence of auto-correlation explaining short run movements in the EU CRC price.

However, after controlling for the instantaneous and lagged effects of US prices on EU prices, other variables tended to show little statistical significance in explaining monthly movements in EU CRC prices. Most importantly, there was very little evidence that contemporaneous movements in the price of EU carbon is economically or statistically different from zero. This result was consistent across all of the models, including the estimation of the co-integrating equation itself, the various VECM specifications, the VAR estimation in first differences, and the OLS model. One-month lagged changes in carbon prices also showed little evidence of being drivers of EU prices. *Overall, we therefore found no evidence to support the hypothesis that EU steel producers pass on the opportunity cost of carbon in their EU product prices.*

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<sup>52</sup> Note that in the presence of non-stationary data this econometric approach is not necessarily unbiased. However, such estimations can be used as a way to explore characteristics of the underlying data, and was intended as such here.

**Table 4.3 CRC steel price regression results**

<b>Parameter</b>	<b>Co-integration Equation</b>	<b>VECM (1)</b>	<b>VECM (2)</b>	<b>VECM (3)</b>	<b>VAR</b>	<b>OLS</b>
<i>CONSTANT</i> (S.E.)	90.6*** (34.2)	-.051 (3.39)	-.109 (3.43)	10.2 (10.4)	-7.33 (54.1)	-950*** (231)
<i>PUS</i>	.853*** (.22)					
<i>PCO2</i>						-.812 (1.14)
<i>PCOAL</i>						4.82*** (.431)
<i>PLABOUR</i>						791*** (261)
<i>PORE</i>						8.68 (37.3)
<i>INDPROD</i>						4.82*** (1.52)
<i>ERROR COR. TERM</i>		-.136*** (.062)	-.126*** (.057)	-.199*** (.038)		
$\Delta PUS$		.694*** (.062)	.682*** (.064)	.545*** (.081)	.473*** (.086)	
$\Delta PCO2$		.149 (1.16)	.116 (1.16)	.138 (1.11)	.266 (1.14)	
$\Delta PCOAL$				.327 (.462)	.738* (.455)	
$\Delta INDPROD$				13.8 (11.7)	.527 (2.07)	
<i>L.ΔPEU</i>				.507*** (.11)	.291*** (.100)	.390*** (.161)
<i>L.ΔPUS</i>					-.005 (.099)	
<i>L.ΔPCO2</i>			1.03 (1.18)	.434 (1.23)	.503 (1.22)	
<i>L.ΔPCOAL</i>					.238 (.401)	
<i>L.ΔINDPROD</i>					-.395 (1.96)	
R-sq	.81	.58	.58	.75	.64	.77
N	97	96	95	97	97	97

\*\*\*Statistically significant at 99%, \*\*Statistically significant at 95%, \*Statistically significant at 90%. Standard errors in parentheses.

As shown in Table 4.3, our results consistently failed to find any evidence that CO2 prices are passed on EU steel prices either in the short or long run. This was true in the co-integrating equations reflecting the long run (see Appendix). It was also untrue of the various VECMs that were estimated, using a variety of control variables and lagged variables to test for the robustness of the results. As a further robustness check, we also estimated a VAR and an OLS equation. These also revealed no evidence of the significance of CO2 prices to EU CRC price formation.

The results strongly suggested that, at monthly intervals, most of the variation in EU CRC prices during the sample period can be explained either by a) contemporaneous fluctuations in international (here proxied by US) CRC prices, or b) by EU price movements to adjust to differences between EU and international prices. After accounting for these effects, none of the other possible price fundamentals that we identified – e.g. coal prices, iron ore prices, industrial production activity, or for that matter CO2 prices – proved to be statistically significant.

*Taken together these facts strongly suggest that EU prices are almost entirely determined by global price fundamentals and thus that the EU is a price taker rather than a price setter in global commodity steel markets.* Given the size of the EU market – which accounts for only around 10% of global production (WSA, 2016) – this result is also consistent with the stylized facts that we know about this market.

*The fact that the EU is a price taker in global commodity steel markets would in turn suggest an explanation as to why the EU does not pass on CO2 opportunity costs in its product prices.* If it did so, the EU would risk becoming significantly out of alignment with competitors prices in global markets. In a competitive international market such as the market for CRC appears to be, this would in turn be expected to encourage almost instantaneous market entry. It would also be likely to significantly undermine EU producers' long run capacity to sell their commodity products in either domestic or international markets and to stay in business.

#### **4.5.2.2. Comparison with previous results**

Our results contrast starkly with the result of de Bruyn et al (2010) which suggested that CO2 prices were passed on by EU steel producers in cold rolled coil (CRC) and cold rolled coil (CRC) prices at around 100%. This raises the question of what might explain this difference in results. Unfortunately, we were unable to replicate the results of this study, even when restricting the sample and control variables to the period used by de Bruyn et al (2010). However, given the extensive use of various modelling alternatives and robustness checks to obtain the above results, we remain confident that the results presented here are robust.

### **4.6. Chapter Conclusion**

The results presented above suggest that EU ETS-covered firms in the cement and commodity steel sector *do not* pass on the opportunity cost of freely allocated carbon allowances into their product prices. In comparison with previous results, we believe that these results are a basis for high confidence in the inability and/or unwillingness of firms in these sectors to pass on their costs. The analysis was conducted over relatively large data samples and the absence of clear evidence of the price pass-through was relatively consistent across a range of different models, controls, and subsets of the sample.

The lack of at least partial evidence is all the more striking in the case of the cement and steel sectors, since these sectors have such strong carbon intensity per unit of output. Even if these sectors were passing on their carbon costs, one would expect to see evidence of it. For example, at 25€/tCO<sub>2</sub>, one would expect to see changes in EU steel prices relative of between 40-50€/t crude steel (a roughly 6.6 to 8% change in steel prices. For cement, the CO<sub>2</sub>-linked price fluctuations should be in the order of 15 to 20€/tcement (i.e. between 15 and 40% of total market prices). It seems implausible that firms would be passing on a high share of these costs in reality and that so little evidence would emerge from the data analysis.

We have ventured plausible explanations as to why firms in these sectors may not be passing on the opportunity costs of allowances into their product prices. These include real-world trade-offs between the benefits of short run optimization of pricing and longer term corporate strategy considerations, both in terms of market entry by foreign rivals and possible regulatory response.

If our conclusions are correct, then they suggest that the EU's current approach to mitigating carbon leakage, at least in the cement and steel sectors, has a serious draw-back. Literature exploring decarbonisation pathways of energy-intensive industrial sectors increasingly highlight the essential importance of improving material efficiency through recycling, end use efficiency, product substitution and innovation. If firms in these sectors in the EU are not passing on a significant share of the price of carbon into their product prices, then crucial incentives along the product value chain to pursue these material efficiency actions do not exist.

## 4.7.Chapter Appendix

### 4.7.1. Cement Data Analysis

#### Breusch-Pagan Test for Random Effects

H <sub>0</sub>	Var(u) = 0
Chi2(1)	0.01
Prob > Chi2	0.999
Conclusion	Reject the null hypothesis: reject pooled regression approach.

#### Hausman Test of Consistency of RE estimation

H <sub>0</sub>	Difference in co-efficients non-systematic
Chi2(6)	3.20
Critical Value(6)	12.58
Conclusion	Fail to reject null hypothesis that difference is non-systematic. Random effects therefore consistent

#### Test of Lag Length

	Coefficient (S.E)	Likelihood ratio Test statistic
L1.CEMP	0.061 (0.053)	24.38*
L2.CEMP	-0.012 (0.223)	23.02
L3.CEMP	0.043 (0.052)	22.98

## 4.7.2. Steel Data Analysis

### Unit Root Test (Dickey Fuller)

H0:	There is a unit root
Z(t) test statistic (n=96)	-1.755
10% critical value	-2.582
Conclusion	Fail to reject the null hypothesis. Unit root assumed.

### Johansen tests for cointegration (no of obs 96, Lags 1, constant trend). Regression of PSTEELEU on PSTEELUS

Maximum rank	Trace statistic	5% critical value
0	170.12	124.2
1*	85.03*	94.2*
2	49.3	68.5
Conclusion	<i>There is one co-integration equation.</i>	

### Breusch-Pagan / Cook-Weisberg test for heteroskedasticity in the VECM equation

H0	Constant variance (no evidence of heteroskedasticity)
Chi2(1)	1.19
PRob > Chi2	0.276
Conclusion	Fail to reject null hypothesis

### Tests for lag-length of VAR, results as given of by SBIC, AIC and HQIC methods respectively.

Optimal Lag Length	
0	-
1*	HQIC, SBIC
2	-
3	-
4	AIC



## Conclusion

Several years have elapsed since the work presented in the preceding chapters was begun. Nevertheless, the issue of carbon leakage as a policy problem that needs to be solved still remains highly relevant today. In short, the same old basic political problem still exists: policy makers want a stronger carbon price to help the economy decarbonise, but they need and currently lack a credible policy story for how to do so without carbon leakage.

Thus it was, that on 26 September 2017, the French President, Emmanuel Macron, outlined a vision for the future of Europe and within that speech he noted a stronger carbon price and a border carbon adjustment to protect EU industries from carbon leakage needed to go hand in hand. The policy combination was the same as those of his two French presidential predecessors, who also advocated such an approach: i.e. yes to a higher price, but also carbon border adjustment". Perhaps there is indeed nothing new under the sun. But in any event the problem to which this thesis was reacting and attempted to bring a partial response has clearly not gone away. If anything it has become more urgent with the greater sense of urgency around climate action that now exists in policy circles and in the wake of the Paris Agreement on Climate Change.

So then, where are we with regard to finding the elusive solution to the carbon leakage issue? Unfortunately, in the EU, the answer seems to be: not that far along. Recently, the EU has opened up another round of reform to its carbon market Directive. Insofar as anti-carbon leakage measures for EITE sectors are concerned, the European Commission and dominant elements of the Parliament and Council process has broadly sought to prolong the status quo with minor reforms to appease some of the concerns raised in this and other research and by industry. For instance, in response to the evidence of threshold effects presented by my colleagues and me in Chapter 3, the Commission has proposed essentially the same ex-ante production, benchmark-based free allocation system as before, but with smaller gaps between thresholds. That is, the allocation will shift every 10% drop or increase in activity rather than only at 25 or 50% changes.

This approach is indeed an efficient solution to the specific problem we raised in Chapter 3 of this thesis and in our original paper and was indeed one of the solutions we identified in our original paper. However, in light of the conclusions highlighted in Chapter 4, which is that this approach to allocation effectively implies no price pass-through of the carbon cost to consumers, the approach is therefore also a major problem. As I argue that chapter, long as there is no price pass through of carbon costs into final product prices by EITEs, it is difficult to see how the EU ETS can drive deep decarbonisation in the sector. This is true on the demand side of these materials markets. However it is arguably also true on the supply side, since it is hard to imagine large firms investing in capital intensive low carbon technologies at new plant without the ability to pass on their costs to consumers.

In a sign of some hints at intellectual progress, the EU Parliament put up an amendment to tackle cement sector anti-leakage measures differently, by proposing a border adjustment and full auctioning. This was, however, shot down by EU parliamentarians in the final vote on ETS amendments in November 2016, due to the ongoing political challenges facing BTAs in the international climate discussions. The EU therefore appears to find itself caught between an unsatisfactory system of anti-leakage measures

and an inability to move to a better system unilaterally. To some this would suggest that the EU should perhaps seek to pursue a more diplomacy-based approach to developing a common system of anti-leakage adjustments with its major international trading partners. At present, however, and in the absence of American leadership internationally, such an approach appears to hold more promise as a long-term rather than a short-term strategy.

On top of the specific approach to anti-leakage measures of the EU, however, there is also a bigger and more troubling question for the current approach to decarbonisation of EITE sectors, which is the complete lack of financial sector bankability of the EU ETS carbon price for making investments on any significant scale. To be sure, the current carbon price remains low and next to useless at close to 6 EUR/tCO<sub>2</sub>eq. However, even in the event of a dramatic recovery in prices, this recovery would need to be sustained over a very long time before the banking sector is likely to be willing to place large financial bets on any investment by EITEs that depend on it. There thus remains the question of whether carbon market based instruments are even a plausible approach to driving deep decarbonisation technology into the sector. This is an area that merits further work by the research community, just in case Mr Macron's proposal of a higher carbon price and border carbon adjustment meets the same fate as that of his two presidential predecessors...

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