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Doctoral Thesis

Environmental Policy and Information: Top-down and Bottom-up Approaches to reach a Sustainable Economy

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A Matteo
Alla mia famiglia
Ai miei amici di sempre
A Zia Sandra
A Gi

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Abstract

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Environmental Policy and Information:
Top-down and Bottom-up Approaches to reach a Sustainable Economy

by Antonia Pacelli

This doctoral thesis in environmental economics investigates the role of policies and information on agents' decision making process. In particular, it explores the scheme of the Emission Trading System (EU ETS) and its effects on economic and environmental outcomes. First, it illustrates the structure of the policy and focuses on the Carbon Border Adjustment Mechanism (CBAM) and the deriving potential scenarios. Then, the thesis goes on an empirical perspective analysing the Italian context and the role of the EU ETS on the emissions' abatement, and exploring the Porter hypothesis. In the Chapter 1 and 2 the agents that take decisions as reaction of the introduction of environmental policies are firms. The third chapter moves the attention to a policy proposal for the European Union, which expands the EU ETS and adds an uniform EU-wide policy too, with the objective to create a EU climate bond for financing the climate investment gap. Lastly, it goes beyond the top-down policies and the importance of creating a bottom-up consensus. This is the reason why the Chapter 4 investigates the role of the introduction of the eco labels on individual and family decisions, considering also children as decision makers. It thus considers as agents individual members of the family and the households as a unit, to understand who takes the decisions in purchasing choices. The overall message of this thesis is that the coordination of top-down and bottom-up approaches might lead to more efficient and faster environmental goal to different types of climate action activities, fostered both by policy makers and individual citizens and communities.

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Chapter 1

The Economics of Carbon Leakage Mitigation Policies

Stefan Ambec[1], Federico Esposito[2] and Antonia Pacelli[3]

In a trade model with endogenous emissions abatement, we investigate the impact of three policy instruments aimed at mitigating carbon leakage: free emission allowances, a Carbon Border Adjustment Mechanism (CBAM), and a CBAM with export rebates. We show that providing free allowances does not alter the incentives to abate carbon emissions, but, instead, fosters the entry of more carbon intensive producers. It "levels the playing field" both domestically and internationally, and may even reverse the carbon leakage. In contrast, a CBAM only levels the playing field domestically, and may lead to an autarky equilibrium. To reverse the carbon leakage, a CBAM must be complemented with export rebates. We further show that a CBAM and export rebates improve welfare for any carbon price, and we identify the optimal share of free allowances with or without a CBAM. Finally, we perform a calibration exercise on cement and steel sectors to simulate the effects of the CBAM recently adopted by the European Union. Our model predicts a scenario with reverse carbon leakage and significant welfare gains for both sectors.

Keywords: Carbon pricing, trade, carbon leakage, CBAM, free allowances, export rebates.
JEL codes: F13, F18, H23, Q52, Q54, Q58.

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1.1 Introduction

Carbon pricing initiatives to tackle climate change have recently been flourishing worldwide. Several jurisdictions have capped greenhouse gas emissions from industrial producers by setting up emission trading schemes, called "cap-and-trade". Examples include the European Union's Emission Trading Scheme (ETS), the Regional Greenhouse Gas Initiative in the northeastern United States, California's and Quebec's joint cap-and-trade program, and China's ETS (Schmalensee and Stavins, 2017, Almond and Zhang, 2021). Companies located in these jurisdictions have to pay for their carbon emissions by buying emission allowances, increasing their production costs, and, therefore, reducing their competitiveness relative to foreign firms. This creates an uneven playing field, with repercussions for international trade flows and the climate. In fact, unilateral carbon pricing may lead to "carbon leakage": since greenhouse gases emitted outside the border of the emission trading market are not capped, the emission reductions induced by the cap-and-trade regulation can be more than offset by an increase of emissions from foreign competitors (see Aichele and Felbermayr, 2012).

Carbon leakage can be mitigated using three policy tools. First, the cost burden due to the carbon price on domestic firms can be lowered with rebates and subsidies based on output, abatement efforts, or emission intensities. Second, the cost of imported goods can be increased with a border charge through a Carbon Border Adjustment Mechanism (CBAM). Third, the cost of exports can be reduced with rebates and subsidies on exported production (Fischer and Fox, 2012). The European Union (EU) has recently been adopting these policies in the context of its Green Deal initiative to tackle climate change. A CBAM entered into its transitional phase in the EU in October 2023 on imports of selected industries (aluminum, cement, hydrogen, fertilizers, iron and steel, and electricity). Imports are charged a carbon tax on their carbon footprint, set equal to the average price of permits traded in the ETS. This CBAM will coexist with free allowances during a transitory period, and will eventually replace them (see European Commission, 2021a).

How do anti-leakage policies impact international competition? How do they affect welfare? What will be the impact of a CBAM on European industries? To answer these questions, we develop a two-country model of international trade in an industry producing an homogeneous good.[4] The carbon emission intensity can be reduced by investing in pollution abatement, which has a cost that is heterogeneous across producers. Carbon emissions are priced with an ETS domestically, at an exogenous price, but not abroad.

We first characterize the equilibrium outcomes to understand how anti-leakage policies improve fair competition, both inside and outside the jurisdiction in which the carbon is priced. We show that by subsidizing output, free allowances level the playing field, not only domestically but also on international markets. A higher share of free allowances can make domestic

[4] The homogeneity assumption allows us to compare the competitiveness of domestic and foreign firms by looking directly at production costs. For industries subject to the CBAM introduced by the EU, it seems a reasonable assumption, as these industries mostly produce raw materials.

firms more competitive abroad, as long as enough resources are invested in dealing with pollution. Such "clean" firms end up exporting to a foreign country, which reverses the leakage problem by lowering the carbon-intensity of products consumed abroad. Since low-emission production at home replaces high-emission production abroad to serve the foreign market, global emissions are reduced, and carbon leakage is negative.

We then analyze the effects of a CBAM. By charging the carbon content of imports, a CBAM levels the playing field domestically: both domestic and foreign firms pay the same cost per unit of CO_2 emitted. This increases the cost of imported products, which reduces imports and, therefore, mitigates carbon leakage. In addition, a CBAM can lead to an autarky equilibrium. This occurs whenever foreign firms are not competitive domestically because of the carbon tariff but, at the same time, domestic firms are not competitive abroad. Nevertheless, a CBAM *alone* does not level the playing field on international markets, as domestic firms exporting abroad are charged for their carbon emissions, while foreign firms are not. In other words, the CBAM reduces and sometimes eliminates carbon leakage, but cannot alone reverse the leakage with exports.[5]

To level the playing field abroad, a CBAM should be complemented with export rebates. By assigning free allowances only on exported output, export rebates have two effects on the equilibrium outcome. First, under the leakage or autarky equilibria, consumers and firms pay the full carbon price (as there are no free allowances), and thus carbon emissions are lower than with free allowances. Second, reverse leakage is more likely because firms have a higher markup per output when they export. In other words, assigning free allowances only to exported output "kills two birds with one stone": it makes firms pay the full cost of their carbon emissions and levels the playing field on international markets.

We then examine the welfare impact of leakage mitigation policies. We show that all allowances should be free without a CBAM, regardless of the equilibrium outcome, or with a CBAM with reverse leakage. Some allowances should be free with a CBAM under carbon leakage if the carbon price is lower than the social cost of carbon. No allowance should be free with a CBAM if carbon is priced at its social cost, except in the case of reverse leakage. We thus highlight another motive for providing free allowances (or subsidizing output): reducing carbon emissions abroad by substituting foreign goods with less carbon-intensive domestic ones on international markets.

Moreover, we show that a CBAM is welfare enhancing for any share of free allowances, and for any carbon price below or equal to the social cost of carbon emissions. Intuitively, with a CBAM, the supply curve in the domestic market reflects the social cost of production, including the carbon cost, at least partially for sub-optimal carbon pricing and fully if carbon is priced at its social cost. The harmful impact of carbon emissions is therefore internalized at least partially or fully, depending on the carbon price. We also show that export rebates further

[5] We also show that free allowances actually *increase* carbon leakage if the carbon border tariff is adjusted by the share of free allowances, as prescribed by the EU legislation for the transition period.

improve welfare by "decarbonating" foreign consumption for different carbon prices that do not exceed the social cost of carbon.

In the last part of our analysis, we calibrate the model to quantify the impact of a CBAM on international trade and welfare. We assume that the home country is the EU, and focus on the two largest manufacturing sectors in which a CBAM is implemented: cement and steel. We use Turkey as the foreign country in the cement sector and Russia in the steel sector, as these are the top exporters to the EU in each industry (among the nations without a formal ETS).[6] We combine publicly available data on production, international trade and emissions to calibrate the model to the year 2019 (before the global COVID-19 pandemic). We also use anonymized plant-level data on emissions intensity (in tons of CO_2 per ton produced) from Italy, made available to us by ISPRA, a public agency that collects environmental data. We use this data to calibrate the abatement cost function and the moments of the distribution of abatement costs.[7]

Our quantitative analysis has three main results. First, increasing the share of free allowances under a CBAM changes the equilibrium outcome from leakage to reverse leakage in both industries. Second, export rebates are more effective in stimulating exports than free allowances, as expected from our theory. Lastly, the welfare gains from a CBAM are large for both sectors and decreasing in the share of free allowances. We also show that these results are generally robust to the calibration used for the abatement cost function and the emission factors.

Related literature Carbon leakage is a concern for both scholars and policymakers. Several studies aim at measuring the magnitude of carbon leakage where carbon is priced. Fischer and Fox, 2012 estimate the impact of a carbon price implemented unilaterally by the US with regard to several energy intensive industries. According to their estimates, a carbon price of \$50 per ton of CO_2 leads to substantial carbon emission leakage rates, ranging from 2% to 58%. Fowlie and Reguant, 2022a analyze the leakage risk across 312 manufacturing sectors in the US and find an average leakage rate of 46% with a carbon price of \$25 per ton of CO_2. Empirically, Aichele and Felbermayr, 2012 find large carbon leakage effects following the implementation of the Kyoto Protocol. Other studies focusing on the EU ETS find limited or no leakage (Bushnell, Chong, and Mansur, 2013; Naegele and Zaklan, 2019; Wagner et al., 2014).

Economists have long advocated for the implementation of border carbon adjustment mechanisms to tackle carbon leakage (see Cosbey et al., 2019, Ambec, 2022, Böhringer et al., 2022 for surveys). Most of the studies investigating the impact of unilateral carbon pricing, CBAM and other anti-leakage policies rely on numerical analysis with computable general equilibrium models (e.g. Branger and Quirion, 2014, Balistreri, Böhringer, and Rutherford, 2018,

[6] For instance, China is also among the top exporters to the EU, but it has a cap-and-trade system in place, which is not consistent with the assumption in our model that foreign firms do not pay a carbon tax.

[7] We conduct our analysis with the anonymized plant level data, adhering to the confidentiality rules set by the ISPRA-DiSES Convention. In particular, our analysis does not reveal any information about any given plant in the dataset.

Balistreri, Kaffine, and Yonezawa, 2019, Böhringer, Schneider, and Asane-Otoo, 2021). They provide quantitative analyses, however, they do not analytically characterize the properties of the equilibrium nor the optimality of anti-leakage policies as we do in this paper.

Earlier works, such as Markusen, 1975, have shown that unilateral carbon pricing can be optimal despite carbon leakage in a two-goods international trade model. Balistreri, Böhringer, and Rutherford, 2018 extended the Markusen model to characterize the optimal carbon tariff with a CBAM. They found that it should be lower than the social cost of carbon because, in their framework, the CBAM increases supply in foreign markets, which lowers the foreign price, increases foreign consumption and, therefore, foreign emissions. We do not have this same effect of a CBAM on foreign prices, because of the assumption of unlimited supply at constant marginal cost in foreign markets. Hence, our carbon tariff is set optimally at the carbon price when the latter equals the social cost of carbon.

Recent studies (Kortum and Weisbach, 2021, Farrokhi and Lashkaripour, 2022, and Weisbach et al., 2023) have identified the optimal policy mix to address carbon leakage using multi-sector models with heterogeneous goods and monopolistic competition à la Melitz (Melitz, 2003). The optimal policy mix involves a carbon tax equal to the social cost of carbon, taxes on imports (based on their carbon content, as in a CBAM), a tax on energy, and export subsidies. In contrast to this literature, we investigate the welfare effects of anti-leakage policy instruments in second-best settings where the optimal policy mix is not implemented. Notably, we extend the welfare analysis to sub-optimal carbon pricing. We show that the CBAM is welfare-enhancing for *any* carbon price, even if it is below the social cost of carbon. In addition, this is the case even when some free allowances are assigned, or when production is subsidized. Moreover, we show that welfare can be improved further if a CBAM is complemented with export rebates for any carbon price below or equal to the social cost of carbon.[8]

Two studies address carbon leakage through the relocation of manufacturing plants outside the jurisdiction in which carbon is priced, a phenomenon sometimes called "pollution off-shoring" (Saussay and Zugravu-Soilita, 2023) or "pollution outsourcing" (Levinson, 2023). Martin et al., 2014 use a calibrated model to estimate the number of allowances that should be freely assigned in the EU ETS in order to achieve a given level of plant relocation. Ahlvik and Liski, 2019 identify carbon policies when firms' relocation costs are private information. Our approach is different, because leakage occurs through international trade, which is absent in both papers. We find out how different carbon leakage mitigation policies affect international trade outcomes. We then characterize the optimal anti-leakage policies depending on the equilibrium within international markets.

[8] It is worth mentioning that our approach differs from Kortum and Weisbach, 2021, Farrokhi and Lashkaripour, 2022, and Weisbach et al., 2023 in at least three dimensions. First, in our paper, the welfare impact of anti-leakage policy instruments is analyzed without any constraints on the foreign country's welfare, nor with strategic interactions among countries. Second, we do not model the energy sector, thus the carbon leakage arises from the reduced competitiveness of domestic firms. Third, we allow for technological change through investment in pollution abatement, while those papers do not.

Our paper builds upon existing partial equilibrium models with trade, particularly Fischer and Fox, 2012 and Fowlie and Reguant, 2022b.[9] Fowlie and Reguant, 2022b characterize and estimate the optimal subsidy in a two-country model, with one representative firm in each country. Similarly, we also characterize the optimal output subsidy with and without a CBAM. However, our formula is different, because, in our model, domestic production is driven by the entry or exit of firms with heterogeneous pollution abatement efforts and emission-intensity.[10] Fischer and Fox, 2012 compare various anti-leakage policies, including carbon border adjustments, in a model with differentiated goods and investment in pollution abatement. In contrast, we characterize the economic outcomes in a model where goods are perfect substitutes, which allows us to compare the competitiveness of firms on both sides of the border.

The rest of the paper proceeds as follows. We first develop a partial equilibrium model to investigate the economic effects of the carbon leakage mitigating policies in Section 2. Next, in Section 3, we perform a welfare analysis and describe the optimal mixes of carbon pricing and free allowances with a CBAM. Section 4 calibrates a parametric version of the model and performs policy simulations. Section 5 concludes.

1.2 A trade model with endogenous emissions abatement

In this section, we develop a partial equilibrium model with two countries (a home country h and an aggregate of the rest of the world, which we call the foreign country f) that can freely trade an homogeneous polluting good. In the home country, carbon emissions are subject to a constant tax. The key feature of the model is that firms choose their optimal investment in carbon emissions abatement, and are heterogeneous in the cost of doing so. In this setting, we characterize the economic and welfare effects of a range of carbon leakage mitigation policies.

1.2.1 Framework

In the home country (h), production is supplied by a continuum of firms of mass 1, each of type θ. Each firm can produce q units of the good with constant marginal cost c_h. Producing the good emits CO_2 with an emission factor (also referred to as emission intensity or carbon footprint) normalized to 1. Firms can reduce the emission factor by a by investing into carbon emissions abatement. The cost of abating carbon emissions is firm specific. Firm of type θ invests $\theta C(a)$ to reach an emission factor of $1-a$, with $0 < a < 1$. We assume $C(a)$ is increasing and strictly convex with $C'(1) = +\infty$, such that production is never fully carbon free. We assume that the firm's abatement cost type θ is distributed according to a density g and a

[9] Böhringer, Fischer, and Rosendahl, 2014 also rely on a partial equilibrium model with trade. They compare the leakage rate and greenhouse emissions induced by several anti-leakage policies in a multi-country setting. However, they do not characterize the equilibrium, nor the optimal anti-leakage policy mix as we do.

[10] Cicala, Hémous, and Olsen, 2022 also model the entry and exit of firms with heterogeneous emission-intensity in their investigation of the impact of the certification process in a CBAM. However, they assume that all firms have same abatement costs, while they are heterogeneous in our setting.

cumulative G, on the range $[\underline{\theta}, \overline{\theta}]$. We assume without loss of generality that $\overline{\theta}$ is larger than all the entry cutoffs we derive throughout our analysis. Examples of abatement strategies include improving energy efficiency or switching to a decarbonated source of energy.[11] We interpret the abatement cost $C(a)$ as a set-up cost for a given production capacity, which is increasing in the emission factor a. This cost is related to the firm's knowledge capital and technological portfolio, including patents, and cannot be transferred or imitated.[12]

The good is also produced in the foreign country (f) with unlimited supply at unit cost c_f and with an emission factor of $\gamma \geq 1$: the production process abroad is at least as carbon intensive as the domestic one. This assumption is consistent with the general lack of carbon pricing that exists outside the EU. While carbon emissions are free in the foreign country, they are priced in the home country at rate $\tau > 0$ per ton of CO_2. Carbon pricing increases the production cost with uncontrolled emissions in the home country from c_h to $c_h + \tau$. We assume that $c_f < c_h + \tau$: carbon pricing makes foreign firms more competitive than domestic ones without pollution abatement.

We assume perfect competition in the sense that firms are price-takers[13], and entry is free.[14] The demand function for the polluting good is $D(p_h)$, decreasing with the price p_h. We denote inverse demand with $P(Q)$ and consumers' surplus with $S(Q) = \int_0^Q P(x)dx$ where Q is the aggregate consumption in the home country.

We now examine three policy tools aimed at addressing carbon leakage: free allowances, a CBAM and a CBAM with export rebates.

1.2.2 Free emissions allowances

We first investigate how providing some emission allowances free-of-charge or subsidizing output affects the economy. In an emission trading scheme, firms receive a share α of free allowances per output with $0 \leq \alpha \leq 1$. Given the price of allowances τ and a benchmark emission factor of 1, getting a share α of allowances for free reduces the cost of carbon pricing from τ to $(1-\alpha)\tau$ per output.[15] The case $\alpha = 0$ corresponds to full carbon pricing, while $\alpha = 1$ means that all allowances are free. By selling the allowances assigned free-of-charge in the

[11] For instance, producing steel with the standard production process of combining iron and coke in a furnace has an emission factor of 2 tons of CO_2 per ton of steel. It can be reduced by recycling steel, by sequestrating and storing the CO_2 emissions from the coke combustion, or using hydrogen combined with hydro or nuclear power instead of coal (see also McKinsey Report).

[12] Note that the model encompasses fully transferable abatement technologies in the specific case of only one type $\theta = \underline{\theta} = \overline{\theta}$, or of very high production capacity q.

[13] Home firms are price-takers even when they are exclusive producers of the good (e.g., when they export), as there is a continuum number of firms, so producers never have control over prices.

[14] Note that, since abatement costs are firm specific, the entry of firms of a given type θ is bounded by the production capacity q. This assumption is without loss of generality, as production capacity can be high enough to fill up domestic demand. Note also that the entry or exit condition would be similar with random abatement, except that it would be ex-post similar to the productivity shock model in Hopenhayn, 1992.

[15] Note that since the number of free allowances is based on past emissions, the firm's current abatement effort a that reduces the emission factor by $1-a$ does not impact them. This grandfathering principle applies to most ETS, including the EU ETS, see Directive 2009/29/EC (European Parliament, 2009) or Martin et al., 2014. Although the current abatement effort would certainly affect the number of allowances a firm would obtain in the subsequent

ETS market, a firm obtains $\alpha\tau$ per output. A share α of free allowances is thus equivalent to a subsidy $\alpha\tau$ per output. Therefore, our analysis encompasses both free allowances in an ETS and output subsidies in any carbon pricing mechanism.[16]

Given α, the profit of firm of type θ with an output market price p and a carbon price τ is:

$$\pi_\alpha(a,\theta) = [p - c_h - \theta C(a) + \alpha\tau - (1-a)\tau]q. \tag{1.1}$$

Each firm θ chooses how much to invest into abatement a to maximize its profit $\pi_\alpha(a,\theta)$. Differentiating $\pi_\alpha(a,\theta)$ with respect to a yields the following first order condition for an interior solution:

$$\theta C'(a) = \tau. \tag{1.2}$$

The firm θ invests in abatement up to equalize the marginal cost of abatement to the marginal benefit (i.e., the price of the carbon emission saved). Investment into abatement is thus driven by the carbon price, regardless of the share of free allowances α. Without loss of generality, we assume that $\theta C'(0) < \tau$ to avoid corner solutions ($a^*(\theta) > 0$ for all θ), and thus the optimal abatement level is:

$$a^*(\theta) = C'^{-1}\left(\frac{\tau}{\theta}\right). \tag{1.3}$$

It is easy to show that as long as some allowances are provided free, some firms can benefit from the carbon pricing through their investment into emission abatement. Indeed, firm θ's optimal profit with 100% free allowances is $\pi_1(a^*(\theta),\theta) = [p - c_h - \theta C(a^*(\theta)) + a^*(\theta)\tau]q$, higher than the unregulated profit $\pi_1(0,\theta) = [p - c_h]q$ as long as $a^*(\theta)\tau > \theta C(a^*(\theta)$. The latter inequality holds by definition of $a^*(\theta)$ whenever $a^*(\theta) > 0$. More generally, a firm of type θ enjoys windfall profits from carbon pricing by receiving a share α of free allowances if $\alpha\tau + (1 - a^*(\theta))\tau > \theta C(a^*(\theta))$: in other words, the net trade of allowances more than offsets abatement costs. Importantly, when production costs are the same in the two countries, $c_h = c_f$, free allowances with abatement make some domestic firms more competitive than foreign firms. In the extreme case where all allowances are free ($\alpha = 1$), home producers are on the same level playing field as foreign ones, that is, they have the same production costs with carbon pricing. However, by abating, home firms can become competitive abroad with their optimal abatement level $a^*(\theta)$.

Although the share of free allowances α does not impact how much a given firm θ invests into abatement $a^*(\theta)$, it determines which firms are profitable depending on their abatement cost type θ. Let us denote $K(\theta,\alpha)$ firm θ's production cost per output net of free allowances α with its optimal management strategy $a^*(\theta)$:

$$K(\theta,\alpha) = c_h + \theta C(a^*(\theta)) + (1 - a^*(\theta) - \alpha)\tau \tag{1.4}$$

phase (e.g., after 2030 for the current 2021-2030 phase), we abstract for the dynamic impact of abatement on future allowances.

[16]Note that with an output subsidy $\alpha\tau$, the parameter α is not bounded by 1. Also, with a carbon tax, α can be interpreted as the share of the tax revenue refunded to firms per unit of output.

We have $\frac{\partial K}{\partial \theta} = C(a^*(\theta)) > 0$ (due to the envelope theorem) and $\frac{\partial K}{\partial \alpha} < 0$: the production cost is increasing with the firm's abatement cost type and decreasing with the share of free allowances. Firm θ produces whenever it is profitable, that is, whenever the selling price p exceeds the unit production cost: $p \geq K(\theta, \alpha)$. The active firm with the highest abatement cost earns zero profit. Let us define the cutoff type $\tilde{\theta}$. It is thus defined by the following zero profit condition (per output):

$$p - K(\tilde{\theta}, \alpha) = 0. \tag{1.5}$$

Since $\frac{\partial K}{\partial \theta} > 0$, all firms of type $\theta < \tilde{\theta}_\alpha$ earn infra-marginal profits per output $p - K(\theta, \alpha) > 0$. They produce up to their production capacity q and, therefore, the aggregate supply is $qG(\tilde{\theta})$.

Before examining the equilibrium outcomes under different trade regimes, we investigate how the cutoff type $\tilde{\theta}$ varies with α and τ. Differentiating (1.5) with respect to α and using (1.3) and (1.4) yields:

$$\frac{d\tilde{\theta}}{d\alpha} = \frac{\tau}{C(a^*(\tilde{\theta}))} > 0. \tag{1.6}$$

Increasing the share of free allowances α (or the output subsidy) increases firms' profits and thus entry. The cutoff type increases and so is total supply $qG(\tilde{\theta})$. Although increasing α does not modify the abatement effort $a^*(\theta)$, now firms with higher abatement cost types θ are supplying the good.

The impact of a higher carbon price on entry and exit is more ambiguous. Differentiating (1.5) with respect to τ and using (1.3) and (1.4), we obtain:

$$\frac{d\tilde{\theta}}{d\tau} = \frac{\alpha - (1 - a^*(\tilde{\theta}))}{C(a^*(\tilde{\theta}))} \tag{1.7}$$

The sign of (1.7) depends on whether the cutoff firm $\tilde{\theta}$ is a net seller or buyer in the allowance market.[17] The firm receives αq allowances while it needs $(1 - a^*(\tilde{\theta}_\alpha))q$ ones to comply with the regulation. If $\alpha < 1 - a^*(\tilde{\theta})$, the firm is short of allowances and must buy the difference $(1 - a^*(\tilde{\theta}) - \alpha)q$. In this case, by (1.7), we have $\frac{d\tilde{\theta}}{d\tau} < 0$. In other words, a higher carbon price reduces the profits of all net buyers including firm $\tilde{\theta}$. The firm's type with zero profit $\tilde{\theta}$ is thus lower (i.e., with lower abatement costs), and home production $qG(\tilde{\theta})$ decreases. In contrast, if $\alpha > 1 - a^*(\tilde{\theta})$, firm $\tilde{\theta}$ is a net seller of allowances, and therefore benefits from carbon pricing. By (1.7), we have $\frac{d\tilde{\theta}}{d\tau} > 0$. A higher carbon price increases firm $\tilde{\theta}$'s profits (as well as the profit of all firms with lower abatement costs $\theta < \tilde{\theta}$ who are also net sellers). It thus favors entry into the industry, and therefore increases production $qG(\tilde{\theta})$ in the home country.

We summarize this comparative statics result in the following Lemma.

[17] If the policy consists of a refunded carbon tax, the sign of (1.7) depends on whether the cutoff firm $\tilde{\theta}$ is a net contributor or beneficiary of the refunded tax system.

Lemma 1 *A higher carbon price favors entry (resp. exit) if the firm with the cut off type $\tilde{\theta}$ is a net seller (resp. buyer) of allowances.*

We now examine the equilibrium outcome under **autarky**. Without trade, the price is determined by domestic demand $p = P(qG(\tilde{\theta}_\alpha))$ which, together with the zero profit condition (1.5), determines the autarky cutoff that we denote $\tilde{\theta}_{A\alpha}$. It is thus defined by the following relationship:

$$P(qG(\tilde{\theta}_{A\alpha})) = K(\tilde{\theta}_{A\alpha}, \alpha). \tag{1.8}$$

Under **free trade**, competition from abroad drives down the equilibrium price to be equal to the foreign production cost. The equilibrium prices in the home and foreign countries are $p_h = p_f = c_f$. Providing that some domestic producers remain competitive at this price,[18] the cutoff firm type $\tilde{\theta}_\alpha$ is defined by replacing p by c_f in (1.5), which leads to:

$$c_f = K(\tilde{\theta}_\alpha, \alpha). \tag{1.9}$$

Domestic supply is $qG(\tilde{\theta}_\alpha)$. The home country imports or exports depending on how the price of the foreign good c_f compares with the autarky price $P(qG(\tilde{\theta}_{A\alpha})$. If it is lower, then demand at this price, $D(c_f)$, exceeds domestic supply under autarky, and the good is imported. Conversely, if c_f is higher than the autarky price, foreign firms are not competitive in the home country, and the difference between domestic production and demand is exported.

We summarize this discussion in the following proposition.

Proposition 1 *For a given share α of free allowances, define the autarky price as $p^{A\alpha} \equiv P(qG(\tilde{\theta}_{A\alpha}))$. The equilibrium outcomes are:*

(a) If $p^{A\alpha} > c_f$: carbon leakage.
Prices are $p_h = c_f = p_f$. Domestic production $qG(\tilde{\theta}_\alpha)$ is lower than consumption $D(c_f)$, the difference being imported.

(b) If $c_f > p^{A\alpha}$: reverse carbon leakage.
Prices are $p_h = c_f = p_f$. Domestic production $qG(\tilde{\theta}_\alpha)$ is higher than consumption $D(c_f)$, the difference being exported.

In the case of no free allowances $\alpha = 0$, since domestic producers cannot compete with foreign ones, the autarky price $p^{A\alpha}$ is strictly higher than the price under free trade $p_h = p_f = c_f$. Hence only case (a) holds. The domestic supply is $qG(\tilde{\theta})$ where the cutoff firm type $\tilde{\theta}$ is such that $\alpha = 0$ in (1.9). The remaining domestic demand $D(c_f) - qG(\tilde{\theta})$ is imported. Emissions related to the imported good are leaked outside of the home country's jurisdiction. In contrast, when a share α of allowances is assigned free-of-charge, domestic production costs are reduced, fostering entry. This translates into an increase of both cutoffs $\tilde{\theta}_{A\alpha}$ (under autarky) and

[18] This occurs if the production cost of the most efficient producer is lower than the price, that is, if $c_h + \underline{\theta}C(a^*(\underline{\theta})) + (1 - a^*(\underline{\theta}) - \alpha)\tau < c_f$.

$\tilde{\theta}_\alpha$ (under free trade) and, thus, an increase of supply. Under autarky, the price $p^{A\alpha}$ decreases, while it remains unchanged at c_f under free trade. Hence, increasing α not only reduces imports (and, therefore emission leakage) by increasing domestic supply, but it may also reverse trade and leakage by shifting the economic outcome from (a) to (b).

Proposition 1 is illustrated in Figure 1.1. The (inverse) demand $P(Q)$ is shown in red. The supply can be found by expressing the cutoff type in terms of domestic demand $Q = qG(\theta)$ into its production cost $K(\theta, \alpha)$. That is, substituting $\theta = G^{-1}(Q/q)$ into $K(\theta, \alpha)$ to obtain $K(G^{-1}(Q/q), \alpha)$. It is shown in blue for $\alpha = 0$ (full carbon pricing) and $\alpha > 0$ (free allowances). Point (A), where home demand and supply curves intersect, representing the equilibrium under autarky and without free allowances. When there is free trade but still no free allowances, the equilibrium shifts to (B): the demand is not fully satisfied by the domestic supply $qG(\tilde{\theta}_0)$ and the difference is imported. Increasing the share α of free allowances moves the supply curve downward from $K(\tilde{\theta}, 0)$ to $K(\tilde{\theta}, \alpha)$ as it makes home firms more competitive. The new equilibrium (C) corresponds to the case in which domestic firms are able to export. Domestic supply $qG(\tilde{\theta}_\alpha)$ exceeds domestic demand $D(c_f)$ and, therefore, the difference $qG(\tilde{\theta}_\alpha) - D(c_f)$ is exported. Hence, under free trade, while the supply curve $K(\tilde{\theta}, 0)$ without free allowances in Figure 1.1 leads to the economic outcome (a) with carbon leakage, where assigning free allowances can move the supply curve down to $K(\tilde{\theta}, \alpha)$ and, therefore, leads to the economic outcome (b) with reverse leakage.

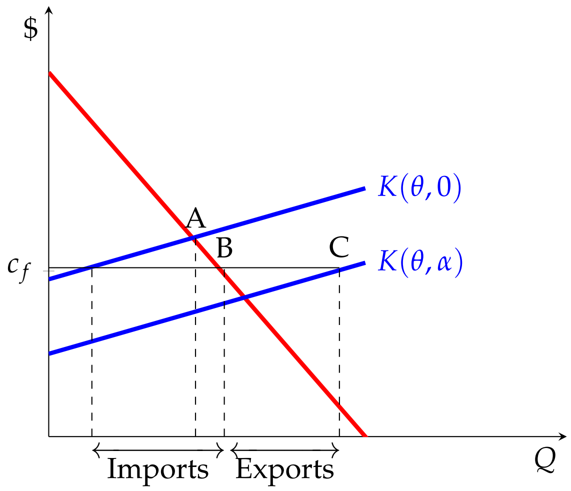

FIGURE 1.1: Equilibria with $\alpha = 0$ (full carbon pricing) and $\alpha > 0$ (free allowances). Point A is the equilibrium under autarky with $\alpha = 0$. Point B is the equilibrium under free trade with $\alpha = 0$. Point C is the equilibrium under free trade with a share $\alpha > 0$ of free allowances.

1.2.3 Carbon Border Adjustment Mechanism

We now analyze the equilibrium outcome with the introduction of a CBAM. The CBAM imposes a tariff on imports based their carbon footprint γ and the carbon price τ. The tariff is $\gamma\tau$ for each good imported in the home country.

With a CBAM, the cost of supplying one unit of good for foreign firms is c_f abroad and $c_f + \gamma\tau$ in the home country. The equilibrium price abroad is $p_f = c_f$. The zero-profit condition that defines the cutoff type $\tilde{\theta}$ depends on which market is relevant for setting the price. If the home country is importing, domestic and foreign firms compete on the home country's market so that the equilibrium price is the highest production cost plus the carbon tariff, $p_h = c_f + \gamma\tau$. In contrast, if the home country exports the good, firms compete outside the home country's borders with an equilibrium price set by foreign firm's production costs on international markets, $p_f = c_f$ (which is unaffected by the carbon price). Hence, we can define a new cutoff type $\tilde{\theta}_{\gamma\alpha}$ whereby the home country imports with a zero profit condition with a domestic price $p_h = c_f + \gamma\tau$ as follows:

$$c_f + \gamma\tau = K(\tilde{\theta}_{\gamma\alpha}, \alpha). \tag{1.10}$$

When instead the home country exports in equilibrium, the cutoff type is defined by the zero-profit condition on foreign markets, that is, with a market price $p_f = c_f$. Hence the cutoff type with exports is the free-trade one denoted $\tilde{\theta}_\alpha$ and defined in (1.9).

The economic outcomes with a CBAM and free allowances are described in the following proposition. The proof is in Appendix 1.5.

Proposition 2 *Under a CBAM with a share α of free allowances, the equilibrium outcomes are:*

(a) If $p^{A\alpha} > c_f + \gamma\tau$: carbon leakage.
Prices are $p_h = c_f + \gamma\tau > p_f = c_f$. Domestic production $qG(\tilde{\theta}_{\gamma\alpha})$ is lower than consumption $D(c_f + \gamma\tau)$, the difference being imported.

(b) If $c_f + \gamma\tau > p^{A\alpha} > c_f$: no carbon leakage.
Prices are $p_h = p^{A\alpha} > p_f = c_f$. The home country supplies its own demand $qG(\tilde{\theta}_{A\alpha})$.

(c) If $c_f > p^{A\alpha}$: reverse carbon leakage.
Prices are $p_h = p_f = c_f$. Domestic production $qG(\tilde{\theta}_\alpha)$ is higher than consumption $D(c_f)$, the difference being exported.

Introducing a CBAM has three distinct effects on the equilibrium of the model. First, it increases the lower bound on the autarky price for case (a) by $\gamma\tau$. This implies that imports and thus carbon leakage are less likely, given the production and abatement costs. Second, it might lead to an autarky equilibrium, which is the new case (b). In fact, starting from case (a) of Proposition 1, the CBAM shuts down imports if $p^{A\alpha} \leq c_f + \gamma\tau$. This "no-trade" outcome occurs for two reasons. On the one hand, foreign firms are no longer competitive domestically because of the CBAM. On the other hand, the share of free allowances α is not sufficiently high to make domestic firms competitive abroad. Producers are fully protected domestically but not competitive enough on international markets. Third, the CBAM increases the domestic price of the good by $\gamma\tau$ in cases (a) and (b). This favors entry as $\tilde{\theta}_{\gamma\alpha} > \tilde{\theta}_{\alpha}$ for any α, which thus increases domestic production compared to case (a) in Proposition 1.[19]

If the CBAM replaces free allowances, the equilibrium outcome described in Proposition is such that $\alpha = 0$. By removing free allowances, both the lower bound for carbon leakage (case a) and the autarky price increase. To see how replacing free allowances with a CBAM modifies the the equilibrium outcome, we illustrate Proposition with $\alpha = 0$ in Figure 1.2 below, and compare it with Figure 1.

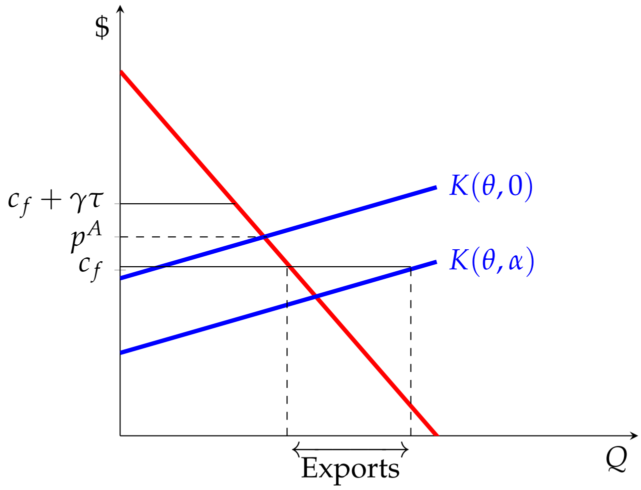

FIGURE 1.2: Equilibria with a CBAM.

Thanks to the CBAM, the full carbon price (i.e., no free allowances $\alpha = 0$) is implemented in equilibrium without carbon leakage in the case graphed in Figure 1.2. It is so because the autarky price with zero free allowances P^A is lower than the cost of imported goods $c_f + \gamma\tau$. The equilibrium outcome is the one described in case (b), namely autarky. The carbon tariff $\gamma\tau$ makes imported goods less competitive than domestic ones. The CBAM eliminated international trade and no carbon emission is leaked.

Carbon emissions do leak if the line $c_f + \gamma\tau$ moves downward below the autarky price p^A (because of lower foreign production cost c_f or emission factor γ). Foreign products are competitive in the domestic market even with a CBAM and, they are therefore imported. Carbon emissions also leak if the supply curve $K(\theta, 0)$ moves upward and crosses the line $c_f + \gamma\tau$ (due to higher domestic production cost c_h or emission abatement costs $\theta C(a^*(\theta))$). Some home producers cannot compete with foreign producers in the domestic market despite the CBAM. Domestic products are replaced by foreign products in the home country.

With a CBAM, free allowances can reverse carbon leakage. It does so by moving the supply curve downward, such that it crosses the demand function (in red) below the horizontal line c_f, as for $K(\theta, \alpha)$ in Figure 1.2. It means that home producers are competitive both in the domestic and foreign markets. They produce at a lower cost than their foreign competitors c_f,

[19]Note that, in case (c) of reverse leakage, the CBAM has no effect on the economy, as nothing is imported. The equilibrium outcome is similar to that in case (b) in Proposition 1.

and are able to fully supply the domestic market, as well as to export. Carbon emissions do not leak outside the home country. On the contrary, home products reduce emissions globally by replacing more carbon intensive foreign products abroad. Carbon leakage is negative.

Moreover, similarly to 1, we now examine how the carbon price impacts entry and exit in the industry with a CBAM. Differentiating (1.10) leads to

$$\frac{d\tilde{\theta}_{\gamma\alpha}}{d\tau} = \frac{\gamma + \alpha - (1 - a^*(\tilde{\theta}_{\gamma\alpha}))}{C(a^*(\tilde{\theta}_{\gamma\alpha}))}. \tag{1.11}$$

Comparing (1.11) with (1.7) shows that $\tilde{\theta}_{\gamma\alpha}$ is more likely to be increasing with τ than $\tilde{\theta}_{\alpha}$. Hence a carbon price increase is more likely to favor entry when a CBAM is implemented. It is so even if the firm of type $\tilde{\theta}_{\gamma\alpha}$ is a net buyer of emission permits. This occurs because home producers benefit from an increase in the carbon price through an increase in the equilibrium price p_h, which might compensate for the net cost of purchasing allowances.

Before moving to analyzing export rebates, we highlight that free allowances are not effective in mitigating carbon leakage with a CBAM if the carbon tariff is adjusted to the share of free allowances, as prescribed in the EU's CBAM proposal during the transition period (Ambec, 2022). All producers, domestic and foreign, will pay the same share of carbon emission $1 - \alpha$ decreasing with the share of free allowance α. The carbon tariff is then set to $\gamma\tau(1 - \alpha)$ during the transition period, and, as α diminishes, it increases up to $\gamma\tau$. Adjusting the CBAM to the share of free allowances more than offsets the reduction of carbon leakage induced by free allowances. It reduces the cost of foreign products by $\gamma\alpha\tau$ while free allowances decrease the cost of domestic products by $\alpha\tau$. With a higher emission factor of foreign products $\gamma > 1$, since $\gamma\alpha\tau > \alpha\tau$, foreign producers obtain a higher cost reduction than domestic ones. Foreign producers become more competitive in the domestic market and thus import more in the home country, which results in more carbon leakage.[20] Carbon leakage turns out to be higher with free allowances. In other words, carbon leakage in the EU would be better addressed by immediately removing free allowances while implementing the CBAM without a transition period.

CBAM and export rebates

We now examine how assigning free allowances only on exported output, a policy called "export rebates", impacts the equilibrium. The share of free allowances is a rebate on the carbon price of the export base. Export rebates with a CBAM causes climate policy to vary in relation to the geographical scope of the market. If the product is sold domestically, the firm has to buy all emissions permits at price τ but is able to sell at a potentially higher price thanks to the

[20] This can be formally shown by noting that adjusting the carbon tariff to free allowances modifies the domestic price with leakage from $c_f + \gamma\tau$ to $c_f + \gamma\tau(1 - \alpha)$ on the left-hand side of (1.10). The supply function $K(\theta, \alpha)$ on the right-hand side is unchanged, the cutoff firm type $\tilde{\theta}_{\gamma\alpha}$ is reduced, as is domestic production $qG(\tilde{\theta}_{\gamma\alpha})$. Since the domestic price is lower, demand increases and imports are higher.

CBAM. If the product is exported, the firm gets a share α of allowances free-of-charge and a price equal to the production cost of its foreign competitors.

Let us consider each of the possible economic outcomes ((a) leakage, (b) no leakage, (c) reverse leakage) with export rebates. Under leakage, since no domestic firms export, no export rebates are provided, and firms buy all of their allowances, so $\alpha = 0$. The cutoff type in the home country market is thus $\tilde{\theta}_\gamma$ defined by equation (1.10). Under no leakage, the same logic applies, because, again, domestic firms do not export. The cutoff type is defined by (1.8) with $\alpha = 0$. In contrast, under reverse leakage, the domestic firms are exporting so they receive export rebates. The zero-profit condition is given by (1.9) so that the cutoff type is $\tilde{\theta}_\alpha$. Proceeding similarly to the proof of Proposition 2, we obtain the following result. The proof is in Appendix 1.5.[21]

Proposition 3 *Define the autarky price when $\alpha = 0$ as $p^A \equiv P(qG(\tilde{\theta}_A))$. With the CBAM and export rebates, the equilibrium outcomes are:*

(a) If $p^A > c_f + \gamma\tau$: carbon leakage.
Prices are $p_h = c_f + \gamma\tau > p_f = c_f$. Domestic production $qG(\tilde{\theta}_\gamma)$ is lower than consumption $D(c_f + \gamma\tau)$, the difference being imported.

(b) If $c_f + \gamma\tau > p^A > c_f + \alpha\tau$: no carbon leakage.
Prices are $p_h = p^A > p_f = c_f$. The home country supplies its own demand $qG(\tilde{\theta}_A)$.

(c) If $c_f + \alpha\tau > p^A$: reverse carbon leakage.
Prices are $p_h = c_f + \gamma\tau > p_f = c_f$. Domestic production $qG(\tilde{\theta}_\alpha)$ is higher than consumption $D(c_f + \gamma\tau)$, the difference being exported.

We can compare Propositions 2 and 3 to understand how export rebates modify the equilibrium outcomes with a CBAM. The cutoff on autarky price p^A that distinguishes between carbon leakage (a) and no carbon leakage (b) is then $c_f + \gamma\tau$ in both Propositions 2 and 3. The carbon leakage and no carbon leakage cases ((a) and (b), respectively) are identical because, since there is no export, the export rebate does not apply. What changes with export rebates is the lower bound on the autarky price P^A, for which the equilibrium involves export and carbon leakage (case (c)). Since this lower bound on P^A increases by $\alpha\tau$, the economy moves from autarky to exports whenever $c_f > p^A > c_f + \alpha\tau$ with export rebates. By exporting, home producers obtain the rebate $\alpha\tau$ in addition to the foreign price c_f, which makes more of them profitable. They are thus able to export and, therefore, to reverse the leakage problem. The export rebate levels the playing field abroad by exempting home producers of a share α of their emission costs. It reduces the gap that the carbon cost paid for supplying the foreign market by $\alpha\tau$ per ton of CO_2 equivalent.

[21] Note that the choice between selling domestically or abroad is straightforward when $\alpha > \gamma$. By selling abroad a firm obtains $p_f + \alpha\tau$ per output while it gets p_h domestically. With equilibrium prices $p_f = c_f$ and $p_h \leq c_f + \gamma\tau$, exporting is more profitable for all firms (regardless of their type θ) when $c_f + \alpha\tau > c_f + \gamma\tau$, that is when $\alpha > \gamma$ with $\tau > 0$. In this case, all firms in the home country export their production, and demand is supplied by foreign firms.

1.3 Welfare analysis

1.3.1 Social welfare with climate cost

In this section, we investigate how free allowances and a CBAM impact social welfare. The negative impact of carbon emissions is embedded into the social welfare through two terms: the social cost of carbon δ and carbon emitted by the sector globally E_W. The social cost of carbon assigns a value to each ton of CO_2 equivalent greenhouse gases. This might differ from the carbon price if the latter is not at its first-best level. By assuming $\tau \leq \delta$, we do not rule out the possibility that carbon is under-priced.

Global emissions E_W are the sum of the domestic and foreign territorial emissions. Denoted E_T, the territorial emissions in the home country are:

$$E_T = q \int_{\underline{\theta}}^{\tilde{\theta}} (1 - a^*(\theta)) dG(\theta). \tag{1.12}$$

To compute the territorial emissions abroad, let D_f be the demand function in the foreign country. Consumption abroad occurs at price $p_f = c_f$ (irrespective of whether the good is produced locally or is imported from the home country). Total production in the foreign country is equal to foreign consumption net of trade, that is, $D_f(c_f) + [D(p_h) - qG(\tilde{\theta})]$. Territorial emissions in the foreign country are thus $\gamma[D_f(c_f) + D(p_h) - qG(\tilde{\theta})]$. Therefore, global emissions are:

$$E_W = q \int_{\underline{\theta}}^{\tilde{\theta}} (1 - a^*(\theta)) dG(\theta) + \gamma[D_f(c_f) + D(p_h) - qG(\tilde{\theta})]. \tag{1.13}$$

The social welfare $\mathcal{W}$ adds up the consumers' surplus net of spending,[22] the producers' profits, transfers (the revenue collected from auctioning allowances and for pricing emissions at the border), net of the social cost of global emissions. Denoting δ the social cost of carbon (each ton of CO_2 being valued δ) and E_W global emissions of the sector, the social welfare without a CBAM is:

$$\begin{aligned}
\mathcal{W} \; = \; & \underbrace{S(D(p_h)) - D(p_h)p_h}_{\text{Consumers' net surplus}} + \underbrace{\int_{\underline{\theta}}^{\tilde{\theta}} \pi_\alpha(a^*(\theta), \theta) dG(\theta)}_{\text{Producers surplus}} \\
& + \underbrace{\int_{\underline{\theta}}^{\tilde{\theta}} q[1 - a^*(\theta) - \alpha]\tau dG(\theta)}_{\text{Auction revenue}} \quad \underbrace{-\delta E_W.}_{\text{Social cost of emissions}}
\end{aligned}$$

With a CBAM, the revenue of collecting the carbon price on imports must be added to the welfare: $\gamma\tau[D(p_h) - qG(\tilde{\theta})]$ with leakage (case (a) of Propositions 2 and 3), and $\gamma\tau D(p_h)$ under reverse leakage and export rebates (case (c) of Proposition 3). Substituting for the profits

[22] By consumers we mean not only final consumers but also producers using the good as an input, for example, car manufacturers. The demand function reflects the private value of the good for all potential clients.

defined in equation (1.1), the auction revenue cancels out with the firms' allowance purchases, so that the welfare with or without a CBAM and reverse leakage simplifies to:

$$\mathcal{W} = S(D(p_h)) - D(p_h)p_h + q\int_{\underline{\theta}}^{\tilde{\theta}} [p_h - c_h - \theta C(a^*(\theta))]dG(\theta) - \delta E_W. \tag{1.14}$$

With a CBAM and carbon leakage, instead we obtain:

$$\begin{aligned} \mathcal{W} \quad = \quad & S(D(p_h)) - D(p_h)p_h + q\int_{\underline{\theta}}^{\tilde{\theta}} [p_h - c_h - \theta C(a^*(\theta))]dG(\theta) \\ & + \gamma\tau[D(p_h) - qG(\tilde{\theta})] - \delta E_W. \end{aligned} \tag{1.15}$$

After the transfers cancel out, the home country's welfare can be decomposed into four terms: the consumer's surplus net of spending, the firms' profit gross of the regulation cost, the revenue for pricing the carbon intensity of imports with the CBAM, and the social impact of carbon emissions.

Before analysing the welfare impact of the different leakage mitigation policies, depending on how emissions are accounted for, we examine the case of no leakage (and thus autarky), in which $D(p_h) = qG(\tilde{\theta})$ and the cutoff type is $\tilde{\theta}_{A\alpha}$ defined in (1.8). Substituting $q\int_{\underline{\theta}}^{\tilde{\theta}} p_h dG(\theta) = p_h qG(\tilde{\theta})$ in (1.15), and using $D(p_h) = qG(\tilde{\theta})$, the welfare in the no-leakage case results in:

$$\mathcal{W} = S(qG(\tilde{\theta}_{A\alpha})) - q\int_{\underline{\theta}}^{\tilde{\theta}_{A\alpha}} [c_h + \theta C(a^*(\theta)) + (1 - a^*(\theta))\delta]dG(\theta) - \delta\gamma D_f(c_f) \tag{1.16}$$

Differentiating W with respect to α, and using (1.3), (1.4) and (1.8), we obtain:

$$\frac{d\mathcal{W}}{d\alpha} = -q[(1 - a^*(\tilde{\theta}))[\delta - \tau] + \alpha\tau]g(\tilde{\theta})\frac{d\tilde{\theta}}{d\alpha}. \tag{1.17}$$

The above first-order condition shows that $\frac{d\mathcal{W}}{d\alpha} < 0$ when $\alpha > 0$ as long as $\tau \leq \delta$: the welfare decreases with the share of free allowances when the carbon price does not exceed the social cost of carbon. Therefore, the optimal share of free allowances is a corner solution $\alpha^* = 0$ for every $\tau \leq \delta$. Unsurprisingly, without carbon leakage, full carbon pricing is optimal for any carbon price not exceeding the social cost of carbon.

1.3.2 Optimal share of free allowances

We examine the impact of free allowances on the home country's welfare. We focus on the leakage or reverse leakage cases of Propositions 1 and 2, in the same way that we have addressed the no-leakage case. We consider the cases with and without a CBAM.

First, without a CBAM, differentiating $\mathcal{W}$ in (1.14) with respect to α, and using (1.4) and (1.9), we obtain:

$$\frac{d\mathcal{W}}{d\alpha} = -q[(1 - a^*(\tilde{\theta}))(\delta - \tau) - \gamma\delta + \alpha\tau]g(\tilde{\theta})\frac{d\tilde{\theta}}{d\alpha}. \tag{1.18}$$

The first term into brackets in (1.18) is the social cost of the cutoff firm $\tilde{\theta}$'s emissions per output that are not internalized. The higher the gap between the carbon price τ and the social cost of carbon δ, the higher this term, which reduces welfare as the share of free allowances increases. This climate cost should be compared to that of foreign production, namely $\gamma\delta$, the second term into brackets. This is because firm $\tilde{\theta}$'s production is replaced by foreign production if firm $\tilde{\theta}$ is not producing, as are the carbon emissions. The welfare decreases with more home production, induced by a higher share of free allowances α, if the climate cost of home production not internalized by the cutoff firm $(1 - a^*(\tilde{\theta}))[\delta - \tau]$ exceeds the climate cost of foreign production.

Second, with a CBAM and leakage (case (a) of Proposition 2), differentiating (1.15 and using (1.4) and (1.10), we obtain:

$$\frac{d\mathcal{W}}{d\alpha} = -q[(1 - a^*(\tilde{\theta}) - \gamma)(\delta - \tau) + \alpha\tau]g(\tilde{\theta})\frac{d\tilde{\theta}}{d\alpha}, \tag{1.19}$$

With a CBAM, the climate cost is partly internalized by foreign firms when importing to the home country. Hence, the welfare impact of increasing home production with a higher share of free allowances depends solely on the difference between the emission intensity of the domestic and foreign products $1 - a^*(\tilde{\theta}) - \gamma$ for the climate cost not internalized $\delta - \tau$. If the cutoff firm produces less carbon intensive products than foreign firms (i.e., if $1 - a^*(\tilde{\theta}) < \gamma$), the welfare can be increased by fostering more home production through free allowances. The magnitude of this welfare increase is the climate cost not internalized by firms $\delta - \tau$.

Lastly, with a CBAM and reverse leakage (case (c) of Proposition 2), differentiating the welfare with respect to α yields (1.18). By increasing free allowances, exports substitute foreign products with home products in international markets. The carbon intensity of those foreign products not being priced means that the carbon impact of this substitution should be evaluated by comparing $\delta - \tau$ with δ. Using (1.18) and (1.19), we prove the following result in Appendix 1.5.

Proposition 4 *All allowances should be free with or without a CBAM under reverse leakage. Some allowances should be free with a CBAM under leakage if $\tau < \delta$ however, none should be free if $\tau = \delta$. Under autarky, no allowance should be free when $\tau \leq \delta$.*

Proposition 4 characterizes the conditions under which free allowances should be part of the carbon mitigation policies. When the domestic market is not protected by a CBAM, assigning allowances free-of-charge turns out to be welfare enhancing, because foreign products with a higher emission-intensity are replaced with domestic products. Thus, global emissions decrease, improving welfare. This substitution effect with free allowances is also welfare enhancing with a CBAM under reverse leakage.

In contrast, with a CBAM and leakage, free allowances improve welfare due to the substitution effect if the climate cost of production is only partly internalized with carbon pricing, that is, if $\tau < \delta$. In contrast, using Pigou pricing $\tau = \delta$, free allowances are no longer optimal. Both consumers and producers (including foreign ones) fully internalize the climate cost of their decisions, and the climate cost δ is embedded into the domestic price.[23]

Note that, in Appendix 1.5, we also investigate to what extent our results hold when $\gamma < 1$ (i.e., when foreign goods have lower carbon emissions than domestic ones). We show that free allowances remain optimal as long as γ is not too low.

Finally, we can proceed similarly to investigate the optimal output subsidy s^* instead of the share of free allowances α^* by setting $s = \alpha\tau$ in (1.18) or (1.19).[24] With or without a CBAM and reverse leakage, the welfare function being concave in s, the optimal subsidy s^* is found by equalizing the left-hand side of (1.18) to zero, which leads to:

$$s^* = \gamma\delta - (1 - a^*(\tilde{\theta}))(\delta - \tau). \tag{1.20}$$

If carbon is priced at its social cost ($\tau = \delta$), then (1.20) reduces to $s^* = \gamma\delta$. The subsidy should ideally compensate for the climate cost of foreign products. If the carbon price is constrained to be lower that the social cost of carbon ($\tau < \delta$), then the subsidy covers the net climate cost that is not internalized.

1.3.3 Welfare impact of the CBAM

We now investigate whether implementing a CBAM improves welfare, conditional on the share of free allowances. We also assess the efficiency of export rebates when a CBAM is implemented. We show the following proposition in Appendix 1.5.

Proposition 5 *A CBAM is welfare-enhancing for any α and $\tau \leq \delta$. Welfare is further improved if the CBAM is complemented with export rebates.*

A CBAM is welfare-enhancing because it makes the domestic market internalize a part, if not all, of the climate externality. Imports are priced at a level closer to their social cost for any carbon price $\tau < \delta$, and at their social cost when $\tau = \delta$. Thus, the domestic price incorporates at least part of the climate cost, and the firms that survive to competition are those with the lowest emission factors. On the supply side, production costs are minimized at the industry level given the cost of one ton of CO_2 emitted τ. On the demand side, only consumers who value the good more than the production cost of the less efficient active firm with the carbon price τ receives it. The welfare is maximized when the carbon price reflects its social cost $\tau = \delta$.

[23] Note that without a CBAM, 100% of allowances should be free, even with Pigou carbon pricing, because the climate costs are not internalized by consumers and/or foreign firms.

[24] Note that the term $\frac{d\tilde{\theta}}{d\alpha}$ should be replaced by $\frac{d\tilde{\theta}}{ds} = \frac{1}{C(a^*(\tilde{\theta})}$ which is found by differentiating $c_f = c_h + \tilde{\theta}C(a^*(\tilde{\theta})) + (1 - a^*(\tilde{\theta}))\tau - s$ with respect to s and $\tilde{\theta}$.

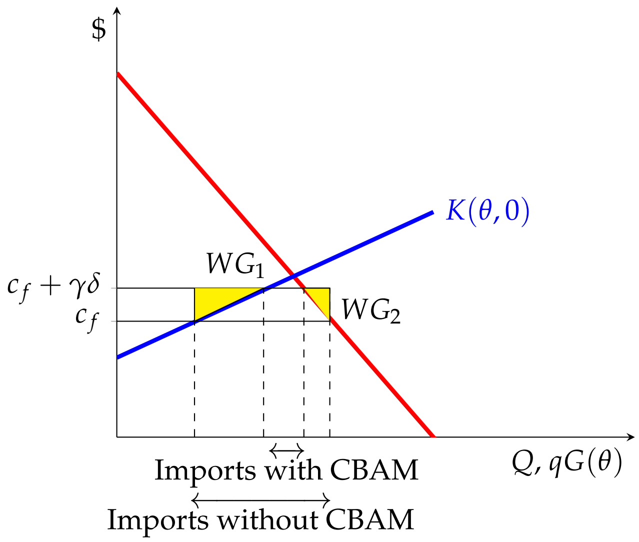

FIGURE 1.3: Welfare gains with CBAM, with $\tau = \delta$.

The welfare gain from implementing a CBAM in case of leakage is shown in Figure 1.3 in the case $\tau = \delta$ and no free allowances. On the supply side, domestic supply $K(\theta, \alpha)$ internalizes the social cost of carbon through carbon pricing, with or without a CBAM. Foreign supply without a CBAM (represented by the line c_f) does not internalize this social cost, unless carbon is priced at the border, in which case the domestic supply is the line $c_f + \gamma\delta$. The area WG_1 is part of the welfare gain from setting up a CBAM. It adds up the difference of social surplus between imports $c_f + \gamma\delta$ and domestic production $K(\theta, \alpha)$ for all imports substituted by domestic production on the left-hand side of the graph. These imports are competitive without a CBAM because their production cost c_f does not include the climate cost $\gamma\delta$. However, they are not optimal because $\gamma\delta$ should be added to the production cost. This is precisely what the CBAM is achieving, causing foreign products to be less competitive.

On the demand side, the equilibrium price with a CBAM c_f is lower than the product's social cost of production $c_f + \gamma\delta$. Consumers whose valuation of the good is in the range between c_f and $c_f + \gamma\delta$ buy the good, while they should not from an efficiency point of view. The area WG_2 is the welfare loss due to this misallocation: the difference between the consumers' valuation of the good and its social cost for all imports that should not be purchased. This loss is avoided by the CBAM, because it increases the equilibrium price at the product's social cost of production $c_f + \gamma\delta$. Overall, the key message of Proposition 5 is that in terms of global emissions, free allowances should be complemented with a CBAM, or replaced by it.

Export rebates further improve welfare because they substitute away carbon-intensive foreign products with low-carbon domestic products in international markets. Unlike free allowances, they do so only when they are effective, that is, under reverse leakage. Furthermore, since export rebates are only applied to exported production, they do not distort the domestic market where carbon is priced.

Lastly, before moving to the quantitative analysis, it is worth discussing two issues related to the real-world implementation of a CBAM. First, note that the Pareto dominance of a CBAM relies on the assumption that the emission factor of foreign products γ is appropriately measured. If this is not the case, the market outcome would be distorted. In practice, measuring the emission intensity of foreign products at the production plant is challenging. For this reason, in the EU's CBAM legislation, a default emission factor is applied at the industry level for products whose carbon footprints are not certified by a reliable third party. Second, although global emissions are the appropriate measure by which to determine the impact of economic activity on the climate, discussions in the policy arena about emissions targets often refer to territorial emissions.[25] In our working paper Ambec, Esposito, and Pacelli, 2023, we show that if territorial emissions only are taken into account, a CBAM actually lowers welfare. This occurs because the CBAM increases domestic production and thus territorial emissions, as well as the domestic price. Those two negative effects are not offset by the higher infra-marginal profits made by the domestic industry with a carbon price at the border.

1.4 Quantitative analysis

We now use our model to investigate the economic impact of carbon leakage mitigation policy tools, with a specific focus on a CBAM. To this end, we first calibrate the model and then simulate several counterfactual scenarios. Given that ours is a partial equilibrium model, we see this exercise as an helpful illustration of the mechanisms used in our framework, rather than a comprehensive assessment of the effects of these policies on the European economy.

1.4.1 Parametric assumptions

To calibrate our partial equilibrium model, we first impose some parametric assumptions on the abatement cost function $C(a)$, the abatement cost distribution, and the demand function of the representative consumer. In particular, we assume that

$$C(a) = \frac{1-(1-a)^{1-\beta}}{1-\beta}, \tag{1.21}$$

where $\beta > 0$. This functional form implies that the abatement costs are convex: increasing the abatement level a (i.e., the fraction of emissions that is produced with clean energy) raises production costs at a rate that increases with a itself. Using this cost function, the first-order condition (1.2) that determines the optimal abatement level $a^*(\theta)$ for a firm of cost type θ writes:

$$(1-a^*(\theta))^{-\beta} = \frac{\tau}{\theta}, \tag{1.22}$$

[25] For instance, to assess their compliance with the Paris Agreement, countries report their emission inventories to the UNFCCC (see UNFCC). In addition, the EU's goal of reducing emissions by 55% in 2030, compared to 1990, and to become neutral by 2050, refers to territorial net emissions that are computed yearly by the EU.

which leads to an optimal abatement level for firm θ of:

$$a^*(\theta) = 1 - \left(\frac{\tau}{\theta}\right)^{-\frac{1}{\beta}}.$$

We assume $\theta \leq \tau$ to make sure that $a^*(\theta) \geq 0$. We further assume that the inverse of θ (i.e., the abatement productivity) is drawn from a log-normal distribution with mean μ and variance σ^2. Lastly, we assume that consumer preferences are such that in each sector, the inverse demand function is iso-elastic:

$$P = \left(\frac{Q}{A}\right)^{-\frac{1}{\epsilon}} \tag{1.23}$$

where $-\epsilon$ is the demand elasticity, Q is the sectoral demand, and A is an exogenous demand shifter. We assume that foreign consumers have the same demand function.

1.4.2 Model Calibration

We calibrate the model to 2019, the latest year before the COVID-19 pandemic impacted the world. We consider two manufacturing sectors that are the target of a CBAM proposed by the EU: cement and steel.[26] We assume that the home country in our model is the EU, while the foreign country is the top exporter to the EU in each sector. Specifically, we use Russia as the foreign country for steel, as Russia was the top exporter of these products to the EU in 2019 (according to trade data from UN Comtrade), among the countries that do not have a cap-and-trade system in place. We use Turkey as the foreign country for cement.

We set τ to €25, the average price of carbon in 2019 in the ETS (European Court of Auditors, 2020). We obtain the average share of free allowances using data from the ETS (see EU ETS). The resulting α_s are close to 1, showing that emissions abatement is heavily subsidized in both sectors. For our simulations, we relax the normalization that the domestic emission rate is 1. Instead, we use estimates from the environmental and engineering literature on the sectoral average emission rates (tons of CO_2 emitted for each ton produced) in EU, Russia and Turkey.[27] We set the sectoral demand elasticities ϵ_s equal to previous estimates in the literature.[28]

We then turn to the estimation of the firms' technology parameters. To this end, we use plant-level data on emissions intensity from Italy, made available to us by ISPRA, a public agency that collects environmental data.[29] We use this data to compute the emission intensity for each

[26] The aluminum, electricity and fertilizers sectors are also targets of the proposal, but the lack of comprehensive data prevents us from including them in our analysis.

[27] Estimates for average emission rates in the EU are obtained from Global Cement and Concrete Association, 2022 and Wörtler et al., 2013. Foreign sectoral average emission rates are based on Turkish estimates for cement (Maratou, 2021) and global estimates for steel (World Steel Association, 2020).

[28] Demand elasticity estimates are from Fowlie, Reguant, and Ryan, 2016 for cement and Reinaud, 2005 for steel. Note that these estimates are taken from the environmental literature, and are lower than the typical estimates from the trade literature (see e.g., Caliendo and Parro, 2015 and Adão, Arkolakis, and Esposito, 2019).

[29] We gratefully obtained the data thanks to a partnership between the Department of Economic and Statistical Sciences of the University of Naples Federico II and the Superior Institute of Environmental Protection and Research (ISPRA).

Italian plant (in tons of CO_2 per ton produced). We use this data set to calibrate the convexity parameter β and the mean and variance of the distribution of the abatement cost θ. To this end, we use the first-order condition (1.22) for the average firm with cost type $E[\theta]$. After normalizing the average abatement cost to 1, we obtain a simple expression linking emissions $e^*(\theta) = 1 - a^*(\theta)$ for all types θ to the carbon price τ:

$$E\left[(1 - a^*(\theta))^{-\beta}\right] = \tau. \tag{1.24}$$

To estimate β, we use the observed emissions per output e_i for all plants i and the observed carbon price τ, and minimize the following function:

$$\beta = argmin\left\{\frac{1}{F}\sum_i e_i^{-\beta} - \tau\right\}, \tag{1.25}$$

where F is the number of plants in our Italian sample (85 in 2019). Our results show that $\hat{\beta} = 1.6$. By inverting the FOC above, we then back out the abatement cost type for manufacturing plant i:

$$\theta_i = \frac{\tau}{e_i^{-\hat{\beta}}}. \tag{1.26}$$

Using the cost types θ_i from (1.26), and assuming that the productivities (the inverse of θ) are drawn from a log-normal distribution, we estimate the mean and variance to be $\mu = -0.96$ and $\sigma^2 = 1.91$, respectively.[30] We obtain the production capacity q_s as the average quantity produced (expressed in tons) across all plants in each sector within the EU.[31] We calibrate the foreign marginal cost, $c_{f,s}$, using the assumption of perfect competition maintained in our model, which implies that the observed import prices should be equal to the foreign marginal cost of production. We use data on unit values per ton from CEPII and compute the average FOB prices of the imports of EU from Russia and Turkey. We then multiply these import prices by the tariffs imposed by the EU on these goods, which we downloaded from the World Bank WITS dataset, to obtain the foreign price $p_{f,s}$.[32]

We calibrate the domestic marginal costs of production by exploiting the fact that the home country (i.e., the EU) in 2019 was a net importer from the foreign country (i.e., either Russia or Turkey) in the two sectors considered in our analysis. Through the lens of our model, this means that for all the domestic producers, in equation (1.1), the equilibrium price is equal to the foreign price $p_{f,s}$. We normalize the profits of the marginal entrant (i.e., a firm with abatement level $a = 0$), in equation (1.1) to 0. Then, since the marginal cost of production, $c_{h,s}$

[30] The average of a log-normal distribution, with mean μ and variance σ^2, is $A = e^{\mu+\sigma^2/2}$, while its variance equals $V = \left(e^{\sigma^2} - 1\right) e^{2\mu+\sigma^2}$. Using the fact that the average of the implied productivities $1/\theta$ is $A = 1$, and that the observed variance is $V = 5.75$, we find $\sigma^2 = ln\left(\frac{V}{A^2} + 1\right) = 1.91$ and $\mu = ln(A) - \sigma^2/2 = -0.96$.

[31] Sources for quantity produced and number of plants by sector are: for cement, Cembureau, 2019 and Cemnet; for steel, European Commission, 2021b and BoldData.

[32] The average tariffs were very low in 2019, being 0 and 0.28 percent for cement and steel, respectively.

is the same across all firms, we can invert equation (1.1) for the marginal entrant in each sector and find $c_{h,s}$.[33]

Lastly, we calibrate the demand shifter A_s, such that our model matches the observed import ratio (defined as imports divided by production) of the EU from the top exporter in each sector. In our model, when the home country is an importer, the import ratio equals:

$$Imp_s = \frac{Demand_s - Production_s}{Production_s} = \frac{A_s \left(p_{f,s}\right)^{-\epsilon_s} - q_s(1 - G(\tilde{\theta}_s))}{q_s(1 - G(\tilde{\theta}_s))},$$

where $\tilde{\theta}_s$ solves the zero-profit condition under free-trade:

$$p_{f,s} + \alpha_s \tau = c_{h,s} + \tilde{\theta}_s \frac{1 - \left(\frac{\tau}{\tilde{\theta}_s}\right)^{\frac{\beta-1}{\beta}}}{1 - \beta} + \left(\frac{\tau}{\tilde{\theta}_s}\right)^{-\frac{1}{\beta}} \tau.$$

We combine the trade data from UN Comtrade with production data from UNIDO to compute the import ratio in 2019 for each sector, and find the demand shifter A_s, such that the model matches the data. Table 1 below reports the relevant parameters by sector.

	Cement	Iron & Steel
Carbon price (τ)	25	25
Share of free allowances (α)	0.99	0.98
Domestic emission rate	0.84	1.29
Foreign emission rate	0.86	1.83
Demand elasticity (ϵ)	-2	-0.9
Convexity parameter (β)	1.60	1.60
Average log-productivity (μ)	-0.96	-0.96
Variance log-productivity (σ^2)	1.91	1.91
Average capacity (q), in thous.	450	0.36
Foreign price (p_f)	185	2,406
Domestic cost (c_h)	185	2,405

TABLE 1.1: Parameters

We discuss the robustness of our quantitative results with respect to the calibrated parameters in Appendix 1.5.

1.4.3 The effects of carbon leakage mitigation policies

We now use the calibrated model to examine the impact of a CBAM, free allowances, and export rebates on trade equilibrium and welfare.

[33] Note that all other entrants make positive profits, because they optimally abate emissions (depending on their heterogeneous abatement efficiency), but they all have the same marginal cost $c_{h,s}$.

Figure 1.4 considers the scenario where the cost of carbon is set to €162, which is the most recent estimate of the social cost of carbon.[34] For each sector, the figure plots the autarky price, the foreign price, and the foreign price under a CBAM for different values for the share of free allowances, α. Without a CBAM, in the cement sector (left panel) an increase in the share of free allowances lowers the autarky price. With low values of α, the autarky price is larger than the foreign price, and thus the home country imports in equilibrium (as in Proposition 1). With high values of α, instead, the home country exports the good. The introduction of a CBAM raises the price of foreign products (foreign price plus carbon tariff) above the autarky one, implying that the home country does not trade in equilibrium when α is low, as the autarky price lies between the foreign price and the foreign price plus the carbon tariff, consistent with Proposition 2. When the share of free allowances is sufficiently high (60%), the home economy switches to exporting, as the autarky price is lower than the foreign price. In the steel sector, a similar pattern emerges, however, the economy switches to exporting only when the share of free allowances is above 80%.

Interestingly, the minimal share of free allowances that is necessary to switch the equilibrium to reverse leakage is higher the lower the carbon price. As shown in Figure 1.9 in Appendix 1.5, when the carbon price is €120, the minimum α that implies exporting is 70% for cement and 90% for steel; when τ is €80 instead, it becomes 80% for cement and 98% for steel. Therefore, when the carbon price is higher, the home country is more likely to export.

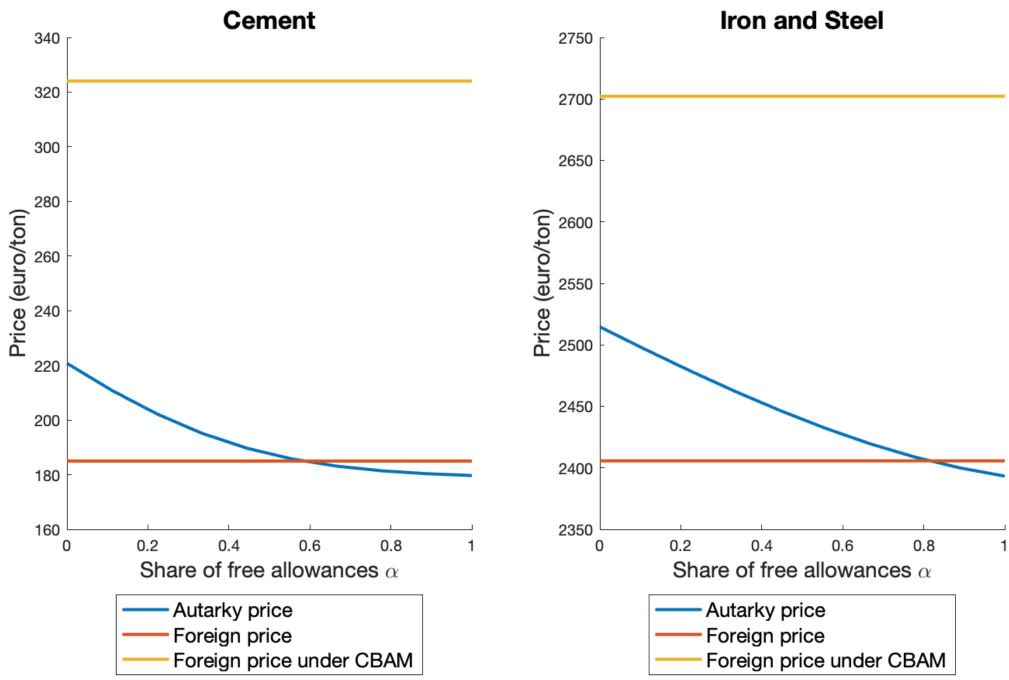

FIGURE 1.4: Equilibrium prices with CBAM, with $\tau = 162$.

[34]The preferred estimate of the social cost of carbon in Rennert et al., 2022 is \$185 in 2020 US dollars. Using the average exchange rate in 2020 between the euro and the dollar, we obtain \$185/1.1422= €162.

The fact that a higher cost of carbon τ increases exports may seem counter-intuitive, as one would expect that a higher cost of carbon increases production costs and thus lowers production. However, in Lemma 1, we have shown that a higher cost of carbon may turn beneficial for domestic producers, if the marginal entrant is a "net seller of allowances," which occurs whenever $\alpha > 1 - a^*(\tilde{\theta}_\alpha)$. To show this mechanism more demonstrably, in Figure 1.10 in Appendix 1.5, we plot the emission intensity of the marginal (or cutoff) entrant, $1 - a^*(\tilde{\theta}_\alpha)$; that is, the firm with abatement cost θ equal to the entry cutoff $\tilde{\theta}_\alpha$, against the share of free allowances α. When α is higher than $1 - a^*(\tilde{\theta}_\alpha)$, which occurs to the right of the 45-degree line, the marginal entrant is a "net seller of allowances." In such a case, increasing τ increases production and, if α is sufficiently high, the home country exports.

We next examine the economic impact of a CBAM combined with export rebates. In Figure 1.5, we display the equilibrium prices with a CBAM when the allowances are granted only to exports. In this scenario, the price schedules differ from when the allowances are given to any output. First, as shown in Proposition 3, the autarky price is found with $\alpha = 0$, and the relevant threshold that switches the equilibrium between autarky and export is now the foreign price c_f plus the export rebate $\alpha\tau$. It is increasing with α and, therefore, the red line is now upward sloping. Second, the autarky price does not depend on α, as domestic production does not grant free allowances. Hence, the autarky price line is now flat. Interestingly, both sectors never import in equilibrium, and switch from autarky to exporting at a lower α compared to the counterfactual in Figure 1.4. This suggests that export rebates are more effective in stimulating exports than production rebates, consistent with Proposition 3.

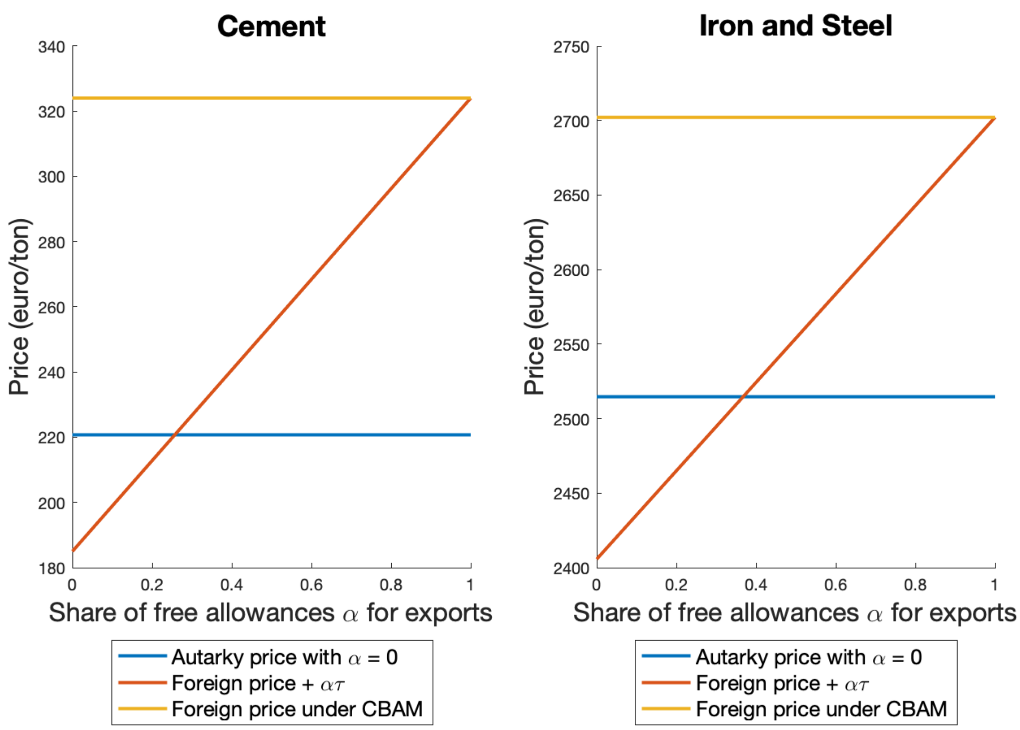

FIGURE 1.5: Equilibrium with CBAM and export rebates, with $\tau = 162$.

1.4.4 Welfare analysis

We now turn to the analysis of the effects of a CBAM on total emissions and welfare. Throughout the section, we set the carbon price to €162, as before, which is the social cost of carbon. In Figure 1.6, we plot both the territorial emissions, using the expression in equation (1.12), and the global emissions, as in equation (1.13). Two patterns emerge in both sectors. First, territorial emissions increase with the share of free allowances, because they foster production by lowering costs, and thus raising carbon emissions. This is very similar to what occurs in a scenario without a CBAM, as shown in Figure 1.11 in Appendix 1.5. In contrast, global emissions first increase with α, but then decrease when the share of free allowances is sufficiently high. This occurs because, as α gets larger, the home country exports the good abroad, as previously shown in Figure 1.4. Following this, the high-carbon emissions of foreign producers are replaced by low-carbon emissions of domestic producers, reducing global emissions, and thus carbon leakage. This differs to what occurs without a CBAM, as Figure 1.11 highlights how free allowances always significantly reduce global emissions, even when α is lower than 1.

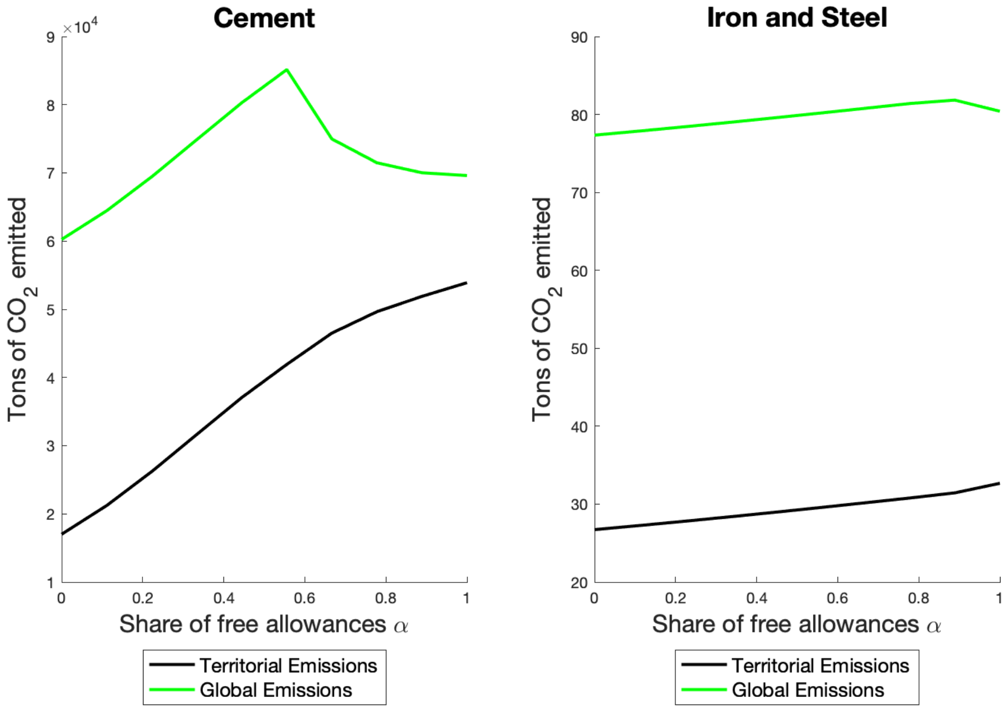

FIGURE 1.6: Emissions with CBAM, with $\tau = 162$.

Next, we look at the welfare effects of a CBAM, separately for each sector, using global emissions.[35] Figure 1.7 plots welfare for different shares of free allowances, normalizing to 1 the

[35]Starting from the demand in equation 1.23, the consumer surplus in sector k can be found as the integral of demand between the willingness to pay, p_0, and the equilibrium price p_h:

$$S_k = \int_{p_h}^{p_0} A_k P^{-\epsilon_k} dP = A_k \frac{1}{1-\epsilon_k}\left((p_0)^{1-\epsilon_k} - (p_h)^{1-\epsilon_k}\right).$$

welfare with $\alpha = 0$. Consistent with Proposition 4, trade-adjusted welfare is decreasing in the share of free allowances if the domestic economy is under autarky, as both sectors are for low levels of α. This is because under autarky the social optimum is attained with $\alpha = 0$, and any $\alpha > 0$ leads to over-production, and thus to an autarky price that is too low. In contrast, when the home country exports the good, giving more free allowances is beneficial, and welfare is increasing in α. This occurs because any extra production generated by a more generous subsidy is absorbed by the foreign country, without any negative effect on the export price (which always equals c_f).

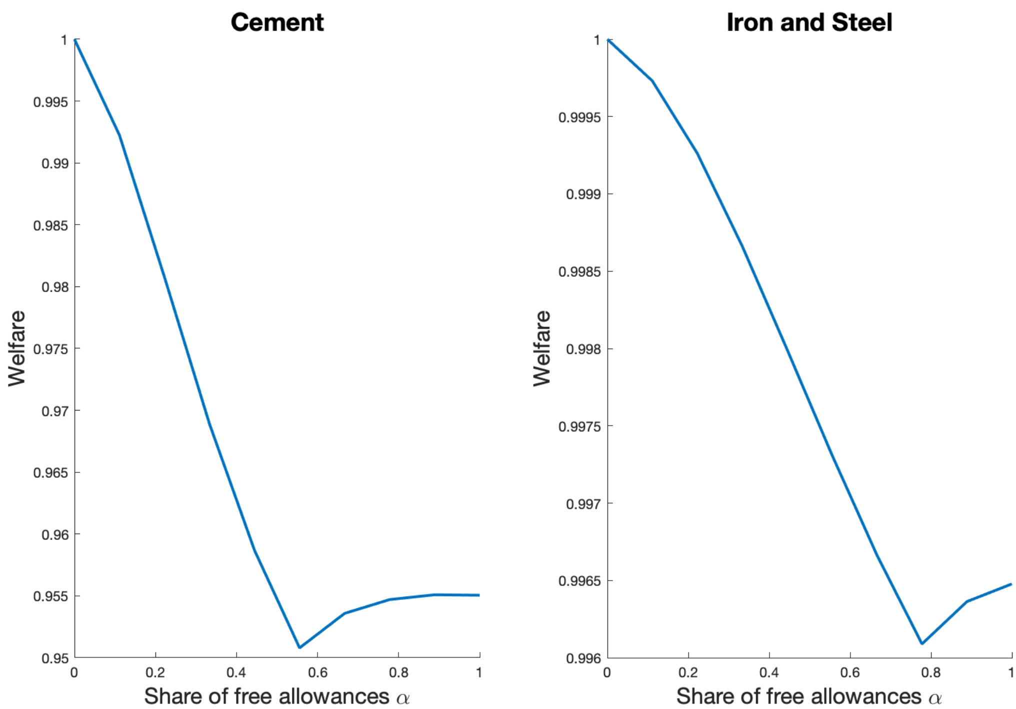

FIGURE 1.7: Welfare with CBAM, $\tau = \delta = 162$.

Finally, in Figure 1.8, we show that welfare with a CBAM is always higher than or equal to welfare without a CBAM, in both sectors.[36] In addition, welfare without a CBAM is always increasing in α, as predicted by Proposition 4. This occurs because giving more free allowances, when $\tau = \delta$, increases production but penalizes the resulting higher emissions with the appropriate social cost. Note that, for low levels of α, the economy is under carbon leakage without a

Note that the lowest quantity that can be consumed is 1, so the willingness to pay is $p_0 = (A_k)^{\frac{1}{\epsilon_k}}$. Replacing it into the above, we get the surplus in sector k:

$$S_k = \frac{A_k}{1-\epsilon_k}\left((A_k)^{\frac{1-\epsilon_k}{\epsilon_k}} - (p_h)^{1-\epsilon_k}\right).$$

We then use equations (1.14) or (1.15) to compute sectoral welfare.

[36]We again normalize to 1 the welfare with CBAM when $\alpha = 0$. Note that in our exercise, we are computing the welfare gains from a CBAM by simply comparing the welfare in the two equilibria. Thus, we are not using a sufficient statistics approach that conditions on observables, as is often seen in the international trade literature (see e.g., Arkolakis, Costinot, and Rodríguez-Clare, 2012 and Esposito, 2020).

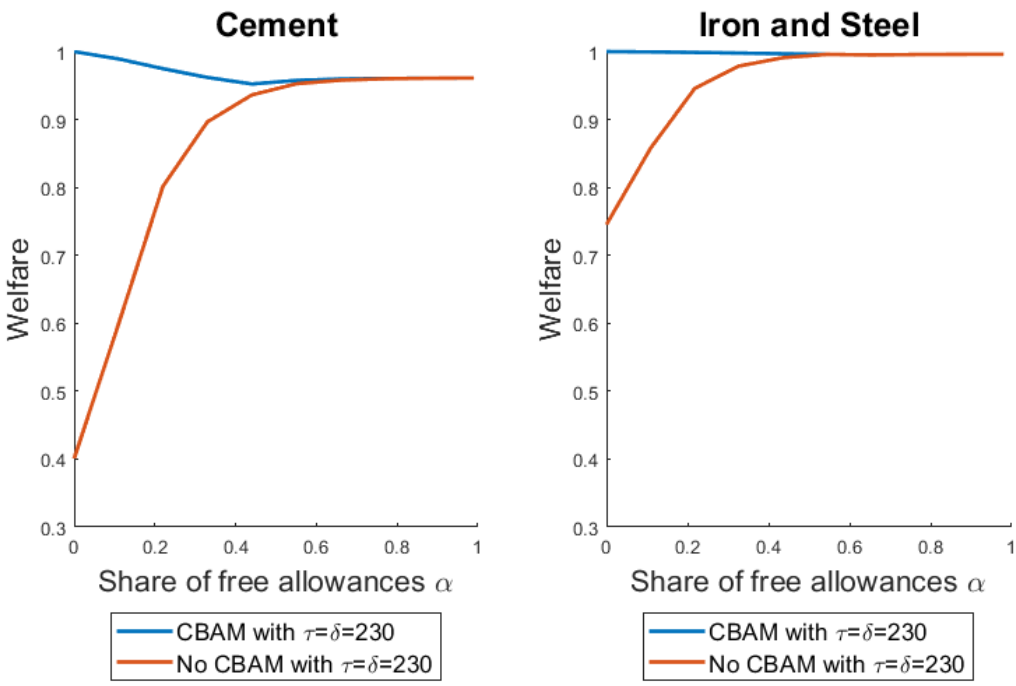

FIGURE 1.8: Welfare with and without CBAM, $\tau = \delta = 162$.

CBAM and in autarky with a CBAM, and welfare with a CBAM is strictly larger than without (as in case (b) of the proof of Proposition 5 in Appendix 1.5). Instead, when α is high, there is reverse leakage both with and without a CBAM (case (c)in the proof of Proposition 5). In this case, welfare is the same with or without a CBAM because the equilibrium outcomes are the same. The CBAM is ineffective because no good is imported and the domestic price is the foreign price. Overall, welfare gains from a CBAM are large and decreasing in the share of free allowances. They range between $0 - 85\%$ for cement and $0 - 19\%$ for steel.

1.5 Conclusions

How can carbon leakage driven by international trade be limited? Should firms be exempt from paying their emission permits, or should the carbon content of imports be taxed with a CBAM? What are the impacts of these leakage mitigation policies? We provide answers to these questions both analytically and quantitatively with a partial equilibrium model calibrated with European data. Although both free allowances and output subsidies are distorted under autarky, they improve welfare in an open economy. By preserving the competitiveness of less carbon-intensive firms, both policies reduce the emission factor of products in the domestic market if the country imports and internationally if it exports. A CBAM does not assist the export process (i.e., it does not lead to reverse leakage), however, free allowances and export rebates do. Providing free allowances on exports makes the export equilibrium more likely, reducing the emission intensity, not only in the producing country, but also internationally. Furthermore, it increases the welfare of a producing country. A CBAM is welfare-enhancing for different reasons: either because it switches the economic outcome from imports to autarky, or it makes firms (and consumers) pay the entire cost of their carbon emission under imports. Our simulations suggest that the EU would gain substantiallyfrom a CBAM in sectors such as cement and steel.

To conclude, we discuss several important assumptions made in our analysis. First, we analyze carbon leakage mitigation policies, taking the carbon price as exogenous. Studying the choice of the carbon price (or an emission target in an ETS) is beyond the scope of this paper, as it would require us to set up a political economy model. However, our model can still shed light on the effects of an exogenous change in the carbon price. We do that analytically in Appendix 1.5. There, we formally show that the carbon price has three distinct effects on welfare: a price effect, an abatement effect, and an entry/exit effect. An interesting avenue for future research could be to quantify these channels in a setting with an endogenous carbon tax.

Second, our analysis relies on the assumption that each country-sector produces an homogeneous good. This leads to the equilibrium outcome in which the domestic country either imports, exports, or does not trade. However, in reality, even raw products, such as aluminum, cement, or steel, may be differentiated by quality, shape, and brand. This means that within the same sector, some varieties are imported, while others are exported. While our model does not allow for intra-industry trade, it should be clear that what is important for our results is whether the home country is a net importer or exporter in a given sector, rather than the product heterogeneity that may exist within a sector. In the same vein, by focusing on only one sector in partial equilibrium, we abstract for spillover effects across sectors. In particular, we do not model the pass-through on prices along the supply chain of the product (e.g., for inputs such as labor or energy) or on complementary or substitute products (e.g., wood instead of cement in the construction sector). Inter-sectoral spillover effects can be modeled and evaluated using computable general equilibrium models.

Third, by assuming that the good can be supplied internationally with constant marginal cost, we abstract for any effect of anti-leakage policies on the foreign price. With an increasing rather than a flat supply curve in the foreign country, the substitution of foreign products by home products, driven by free allowances and a CBAM, would lower the foreign price. It would also increase consumption abroad and, thus, mitigate the reduction of global emissions through a scale effect. Therefore, the welfare improvement from CBAM will be lower.

Fourth, the emission intensity of foreign products, γ, is exogenous in our model. Nonetheless, foreign firms might be able to reduce γ by investing in pollution abatement, as their domestic competitors do. For instance, as discussed in the EU proposal, this may require the existence of a certification process (as studied in Cicala, Hémous, and Olsen, 2022). Endogenizing γ with foreign investment in abatement would not significantly change our results. Providing that the imports are charged with firm-specific and well-evaluated emission factors, it would cause a

CBAM to be even more attractive by fostering decarbonization abroad. The optimality of free allowances and export rebates with a CBAM should be assessed by comparing the emission factors on both sides of the border, as we explain in Section 3.2. However, this comparison may be challenging to implement in practice and we leave it for future research.

Lastly, our single-sector model does not differentiate between direct and indirect emissions. The EU CBAM mandates the reporting of both direct and indirect emissions per product (scope 2). In contrast, in the EU ETS, only direct emissions (scope 1) from manufacturing plants are accounted for. Emissions from inputs in the production process, such as electricity, are not included. This asymmetry in the scope of emissions between foreign and domestic products is only an issue if indirect emissions from domestic production are not priced. This is generally not the case for electricity production, since thermal power plants have to comply with EU ETS, however, it could be the case for other inputs produced by manufacturing plants exempted from complying with the EU ETS. Investigating this feature of the EU policy would require extending our model to multiple sectors. We think this is an interesting avenue for future research.

Appendix

Proof of Proposition 2

Under the CBAM, autarky (no leakage) is the equilibrium outcome if the domestic price $p_h = P(qG(\tilde{\theta}_{A\alpha}))$ (with $\tilde{\theta}_{A\alpha}$ defined in (1.8)) is lower than the cost of imported products $c_f + \gamma\tau(1-\alpha)$ (to avoid imports) and higher than the foreign price $p_f = c_f$ (to avoid exports). Hence whenever $c_f < P(qG(\tilde{\theta}_{A\alpha})) < c_f + \gamma\tau$, the home country does not trade. Domestic firms supply domestic demand with $qG(\tilde{\theta}_A\alpha)$ units of the good.

If $c_f + \gamma\tau < P(qG(\tilde{\theta}_{A\alpha}))$, foreign products are competitive in the home country with the CBAM which charges $\gamma\tau$ per unit imported. The domestic price equals to the cost of imported products $p_h = c_f + \gamma\tau$. With this price in the home country, the cutoff firm's type is found by replacing $p = c_f + \gamma\tau$ in (1.5), which leads to (1.10) which defines $\tilde{\theta}_{\gamma\alpha}$. Domestic production is thus $qG(\tilde{\theta}_{\gamma\alpha})$. It supplies the home country with $D(c_f + \gamma\tau)$ units of the product, the rest $D(c_f + \gamma\tau) - qG(\tilde{\theta}_{\gamma\alpha})$ being imported.

If $P(qG(\tilde{\theta}_{A\alpha})) < c_f$, home production is competitive abroad. Home firm exports their production which is sold at price $p_h = p_h = c_f$. The cutoff firm's type is now found by replacing $p_h = c_f$ in (1.5), which leads to (1.9) which defines $\tilde{\theta}_\alpha$. Domestic production is $qG(\tilde{\theta}_\alpha)$, from which $D(c_f)$ is consumed domestically, the rest $qG(\tilde{\theta}_\alpha) - D(c_f)$ being exported.

Proof of Proposition 3

With an export rebate of $\alpha\tau$ and a CBAM, autarky (no leakage) is the equilibrium outcome if (i) the domestic price $p_h = P(qG(\tilde{\theta}_A))$, with $\tilde{\theta}_A$ defined in (1.8) with $\alpha = 0$, is lower than the cost of imported products $c_f + \gamma\tau$ (to avoid imports), and (ii) the revenue that domestic producers get per output exported $p_f + \alpha\tau = c_f + \alpha\tau$ is lower than by selling domestically at price $p_h = P(qG(\tilde{\theta}_A))$ (to avoid exports). Hence whenever $c_f + \alpha\tau < P(qG(\tilde{\theta}_{A\alpha})) < c_f + \gamma\tau$, the home country does not trade. Domestic firms supply domestic demand with $qG(\tilde{\theta}_A)$ units of the good.

If $c_f + \gamma\tau < P(qG(\tilde{\theta}_A))$, foreign products are competitive in the home country with the CBAM (which charges $\gamma\tau$ per unit imported). The domestic price equals to the cost of imported products $p_h = c_f + \gamma\tau$. With this price in the home country, the threshold firm's type with the highest abatement cost is found by replacing $p_h = c_f + \gamma\tau$ in (1.5), which leads to (1.10) which defines $\tilde{\theta}_\gamma$ with $\alpha = 0$. Domestic production is thus $qG(\tilde{\theta}_\gamma)$. It supplies the home country with $D(c_f + \gamma\tau)$ units of the product, the remaining demand $D(c_f + \gamma\tau) - qG(\tilde{\theta}_\gamma)$ being imported.

If $P(qG(\tilde{\theta}_A)) < c_f + \alpha\tau$, home producers can export. Their revenue is $c_f + \alpha\tau$ by exporting. If they sell in the home country, they obtain the market price in the home country which is set at the cost of imports products $c_f + \gamma\tau$. Since $P(qG(\tilde{\theta}_A)) < c_f + \gamma\tau$, then $qG(\tilde{\theta}_A) > D(c_f + \gamma\tau)$ and, therefore, the supply from home producers at price $p_h = c_f + \gamma\tau$ yields strictly positive profits. The zero profit condition is therefore met on exports. The cutoff firm is $\tilde{\theta}_\alpha$ defined in (1.9). Production in the home country is thus $qG(\tilde{\theta}_\alpha)$. Demand in the home country at this price is $D(c_f + \alpha\tau)$.

Proof of Proposition 4

First, consider the case without CBAM or with CBAM and reverse leakage. The welfare impact of free allowances $\frac{d\mathcal{W}}{d\alpha}$ is given by (1.18). We show by contradiction that $\frac{d\mathcal{W}}{d\alpha} > 0$ for every $\alpha < 1$. Suppose $\frac{d\mathcal{W}}{d\alpha} \leq 0$ for one α such that $0 < \alpha < 1$ at least. Then the term into bracket on the right-hand side in (1.18) should be weakly positive, which implies $\tau[\alpha - (1 - a^*(\tilde{\theta}))] \geq \delta[\gamma - (1 - a^*(\tilde{\theta}))]$. Since $\tau \leq \delta$, for the former inequality to hold, we must have $\alpha \geq \gamma$, which, combined with $\gamma \geq 1$, yields $\alpha \geq 1$, a contradiction. From $\frac{d\mathcal{W}}{d\alpha} > 0$ for every $\alpha < 1$, we conclude that the welfare increases with α up to $\alpha = 1$. Hence $\alpha^* = 1$.

Second, with a CBAM and leakage whereby $\frac{d\mathcal{W}}{d\alpha}$ is defined in (1.19), we have $\frac{d\mathcal{W}}{d\alpha}|_{\alpha=0} > 0$ if $\delta < \tau$ with $\gamma \geq 1 \geq 1 - a^*(\tilde{\theta})$ as assumed here. Furthermore, substituting $\tau = \delta$ into (1.19) yields:

$$\frac{d\mathcal{W}}{d\alpha} = -q\alpha\delta g(\tilde{\theta})\frac{d\tilde{\theta}}{d\alpha} < 0,$$

for every $\alpha > 0$ so that the welfare is always decreasing with α. Hence $\alpha^* = 0$ with Pigou carbon pricing with a CBAM and leakage.

The case with $\gamma < 1$

We briefly examine the efficiency of free allowances under the alternative assumption $\gamma < 1$. First, without CBAM or under CBAM and reverse leakage, $\frac{d\mathcal{W}}{d\alpha}|_{\alpha=0} > 0$ in (1.18) if $\delta < \tau$ and $\gamma\delta > (1 - a^*(\tilde{\theta}))[\delta - \tau]$. It implies that the welfare increases with α at zero and, therefore, $\alpha^* > 0$. Furthermore, substituting $\tau = \delta$ into (1.18) yields:

$$\frac{d\mathcal{W}}{d\alpha} = q\delta[\gamma - \alpha]g(\tilde{\theta})\frac{d\tilde{\theta}}{d\alpha}. \tag{1.27}$$

Since $\frac{d\mathcal{W}}{d\alpha} > 0$ if $\alpha < \gamma$ and $\frac{d\mathcal{W}}{d\alpha} < 0$ if $\alpha > \gamma$, which implies that $\mathcal{W}$ is increasing with α up to $\alpha = \gamma$ and decreasing with α for $\alpha > \gamma$. It is thus maximized at $\alpha^* = \gamma$.

Second, under CBAM and leakage, $\frac{d\mathcal{W}}{d\alpha}|_{\alpha=0} > 0$ in (1.19) if $\delta < \tau$ and $\gamma > 1 - a^*(\tilde{\theta})$. Hence $\alpha^* > 0$ in this case. If $\tau = \delta$, $\frac{d\mathcal{W}}{d\alpha}$ is given by (1.27) so that $\alpha^* = 0$.

Proof of Proposition 5

We consider in sequence the three cases described in Proposition 1.2.3.

(a) $p^{A\alpha} > c_f + \gamma\tau$: Leakage with and without CBAM.

The welfare without CBAM can be written as:

$$\begin{aligned}\mathcal{W} &= \int_0^{\tilde{\theta}_\alpha} [P(qG(\theta)) - c_h - \theta C(a^*(\theta)) - (1-a^*(\theta))\delta] dG(\theta) \\ &+ \int_{qG(\tilde{\theta}_\alpha)}^{D(c_f)} [P(x) - c_f - \gamma\delta]dx - \delta\gamma D(c_f). \end{aligned} \tag{1.28}$$

The welfare with CBAM under leakage is:

$$\begin{aligned}\mathcal{W} &= \int_0^{\tilde{\theta}_{\gamma\alpha}} [P(qG(\theta)) - c_h - \theta C(a^*(\theta)) - (1-a^*(\theta))\delta] dG(\theta)] \\ &+ \int_{qG(\tilde{\theta}_{\gamma\alpha})}^{D(c_f+\gamma\tau)} [P(x) - c_f - \gamma\delta]dx - \delta\gamma D(c_f). \end{aligned} \tag{1.29}$$

Since $\tilde{\theta}_{\gamma\alpha} > \tilde{\theta}_\alpha$ and $D(c_f) > D(c_f + \gamma\tau)$, the welfare difference with CBAM minus without CBAM (1.29)-(1.28) writes:

$$\begin{aligned}\Delta\mathcal{W} &= \int_{\tilde{\theta}_{\gamma\alpha}}^{\tilde{\theta}_\alpha} \underbrace{[c_f + \gamma\delta - c_h - \theta C(a^*(\theta)) - (1-a^*(\theta))\delta]}_{(i)} dG(\theta)] \\ &- \int_{D(c_f+\gamma\tau}^{D(c_f)} \underbrace{[P(x) - c_f - \gamma\delta]}_{(ii)} dx. \end{aligned} \tag{1.30}$$

First, by (1.9) and because $\theta C(a^*(\theta)) + (1 - a^*(\theta))\tau$ is increasing with θ, we have $c_f - c_h - \theta C(a^*(\theta)) > (1 - a^*(\theta) - \alpha)\tau$. The last inequality implies that (i) in (1.30) is higher than:

$$\gamma\delta - \alpha\tau - (1 - a^*(\theta))[\delta - \tau], \tag{1.31}$$

for every $\theta < \tilde{\theta}_\alpha$. Since $\gamma \geq 1 \geq \alpha$, (1.31) is weakly higher than $[\gamma - (1 - a^*(\theta))] \geq 0$, where the last inequality is due to he fact that $\gamma \geq 1 \geq 1 - a^*(\theta)$ for every θ and $\tau \leq \gamma$. Hence, (i) in (1.30) is strictly positive.

Second, for (ii) in (1.30), remark that $x > D(c_f + \gamma\tau)$ implies $P(x) < c_f + \gamma\tau$ by definition of $D(.) = P^{-1}(.)$ and $D'(.) < 0$. By $\tau \leq \delta$, $P(x) < c_f + \gamma\tau$ implies $P(x) < c_f + \gamma\delta$ for every $x > D(c_f + \gamma\tau)$. Hence the second integral in the right-hand side of (1.30) is strictly negative.

We conclude $\Delta\mathcal{W} > 0$.

(b) $c_f + \gamma\tau > p^{A\alpha} > c_f$: leakage without CBAM and no leakage with CBAM.

The welfare without CBAM is given by (1.28), while the welfare with CBAM under no leakage is given by

$$\mathcal{W} = \int_0^{\tilde{\theta}_{A\alpha}} [P(qG(\theta)) - c_h - \theta C(a^*(\theta)) - (1-a^*(\theta))\delta] dG(\theta). \tag{1.32}$$

The welfare difference with and without CBAM (1.28) minus (1.32) is:

$$\begin{aligned}\Delta\mathcal{W} &= \int_{\tilde{\theta}_{\gamma\alpha}}^{\tilde{\theta}_{A\alpha}} [c_f + \gamma\delta - c_h - \theta C(a^*(\theta)) - (1 - a^*(\theta))\delta] dG(\theta) \\ &\quad + \int_{qG(\tilde{\theta}_{A\alpha})}^{D(c_f+\gamma\tau)} [P(x) - c_f - \gamma\delta] dx.\end{aligned}$$

Proceeding as for leakage case (a) shows that $\Delta\mathcal{W} > 0$.

(c) $c_f > p^{A\alpha}$: reverse leakage with and without CBAM.

The welfare is the same with or without CBAM for any given share of free allowances α because the equilibrium outcomes are the same. The CBAM is ineffective because no good is imported and the domestic price is the foreign price.

As for export rebates, they are effective only is case (c) of Proposition 3, in which case the welfare $\mathcal{W}$ is defined in (1.14). Differentiating (1.14) with respect to α, and using (1.4) and (1.9), yields (1.18). In 1.5 we show that $\frac{d\mathcal{W}}{d\alpha} > 0$ for every $\alpha < 1$. Hence $\mathcal{W}$ increases with export rebates $\alpha > 0$.

Impact of the carbon price

We now investigate the impact of the carbon price on the welfare with carbon leakage, accounting for territorial emissions first, and for the home country's contribution to total emissions next. Differentiating (1.15) with E_T instead of E_W with respect to τ and using (1.2) and (1.5) yields:

$$\begin{aligned}\frac{d\mathcal{W}}{d\tau} &= \underbrace{[qG(\tilde{\theta}) - D(p_h)]\frac{dp_h}{d\tau}}_{\text{Price effect}} \\ &\quad \underbrace{+ q\int_{\underline{\theta}}^{\tilde{\theta}} [\delta - \tau]\frac{da^*(\theta)}{d\tau} dG(\theta)}_{\text{Abatement effect}} \\ &\quad \underbrace{+ q[\tau(1 - a^*(\tilde{\theta}) - \alpha) - \delta(1 - a^*(\tilde{\theta}))]g(\tilde{\theta})\frac{d\tilde{\theta}}{d\tau}}_{\text{Entry or exit effect}}.\end{aligned} \tag{1.33}$$

A marginal increase of the carbon price has three impacts on the welfare. First, a higher carbon price might increase the price of the good p_h (in cases (a) and (b) but not (c)) which impacts positively firm's revenue but negatively consumer's spending. We call this channel the *price effect*. It corresponds to the right-hand term in the first line in (1.33). The price effect is negative if production $qG(\tilde{\theta})$ is lower than consumption $D(p_h)$, that is with imports (case (a)). It is nil under autarky (case (b)) since then $qG(\tilde{\theta}) = D(p_h)$: the increase of the good price is just a transfer from consumers to producers. With exports (case (c)), since $p_h = c_f$ (the domestic price is determined by the international price of the good), $\frac{dp_h}{d\tau} = 0$ so there is no price effect.

Second, pollution abatement improves the welfare by increasing pollution abatement. This *abatement effect* shows up the second line of (1.33). A marginal tax raise increases firm θ's abatement $a^*(\theta)$ by $\frac{da^*(\theta)}{d\tau} = \frac{1}{\theta C''(a^*(\theta))} > 0$, which reduces climate cost by δ while at the same time increases abatement cost by $\tau = \theta C'(a^*(\theta))$, where the last equality is due to (1.2). The abatement effect is nil with Pigou pricing $\tau = \delta$, and positive when carbon is under-priced $\tau < \delta$.

Third, a tax increase varies supply through entry or exit in the home country. We call this impact captured in the last line of (1.33) the *entry or exit effect*. As mentioned before, a higher tax favors entry if the threshold firm $\tilde{\theta}$ is a net seller of allowances (in which case $\frac{d\tilde{\theta}}{d\tau} > 0$), or induces some exists if it is a net buyer (then $\frac{d\tilde{\theta}}{d\tau} < 0$). The term into brackets in the third line of (1.33) is the difference between firm $\tilde{\theta}$'s regulatory cost $\tau(1 - a^*(\tilde{\theta})) - \tau\alpha$ and the climate cost $\delta(1 - a^*(\tilde{\theta}))$ per output. If the two coincide, e.g. under Pigou pricing $\tau = \delta$ and no free allowances $\alpha = 0$, the entry and exit effect is nil because firms internalize correctly the climate costs. Otherwise, the sign of the entry or exit effect depends upon both the difference between the regulatory and climate cost of firm $\tilde{\theta}$'s production, and firm $\tilde{\theta}$'s net position of in the allowance market (buyer or seller). If the regulation cost is too low - because carbon is under-priced $\tau < \delta$ and/or some allowances are free $\alpha > 1$ - then the entry and exit effect is negative when a higher carbon price favors entry, which turns out to be the case if the threshold firm is a net seller of allowances (i.e. if $\alpha > 1 - a^*(\tilde{\theta})$). In contrast, it is positive when a higher carbon price make firms exit the industry, that is if the threshold firm is a net buyer of allowances (i.e. if $\alpha < 1 - a^*(\tilde{\theta})$).

With global emissions E_W in the welfare, differentiating (1.15) with respect to τ and using (1.2) and (1.5) yields:

$$\begin{aligned}\frac{d\mathcal{W}}{d\tau} = & \underbrace{[qG(\tilde{\theta}) - D(p_h) - \delta\gamma D'(p_h)]\frac{dp_h}{d\tau}}_{\text{Price effect}} \\ & \underbrace{+q\int_{\underline{\theta}}^{\tilde{\theta}}[\delta - \tau]\frac{da^*(\theta)}{d\tau}dG(\theta)}_{\text{Abatement effect}} \\ & \underbrace{+q[\tau(1 - a^*(\tilde{\theta})) - \alpha) - \delta(1 - a^*(\tilde{\theta}) - \gamma)]g(\tilde{\theta})\frac{d\tilde{\theta}}{d\tau}}_{\text{Entry or exit effect}}.\end{aligned} \tag{1.34}$$

Compared to the case with territorial emissions in (1.33), the above relationship differs in two ways. First, the price effect takes into account the social gain of reduced emissions from lower consumption in the home country, i.e. the last term into brackets of the right-and term in the first line of (1.34). An increase of p_h with a marginally higher τ decreases demand by $-D'(p_h)$ in cases (a) (with imports), which reduces emissions by γ and has social value δ. The marginal climate gain from the price increase with imports is therefore $-\delta\gamma D'(p_h) > 0$. Second, the entry or exit effect measures the carbon impact of the threshold firm's output relative to the foreign alternative rather in absolute term, i.e. with $1 - a^*(\theta) - \gamma$ rather than $1 - a^*(\theta)$. It therefore lower and can even be even be positive if $1 - a^*(\theta) < \gamma$, in which case firm $\tilde{\theta}$'s production improves the welfare by replacing more carbon-intensive foreign products.

Additional Figures

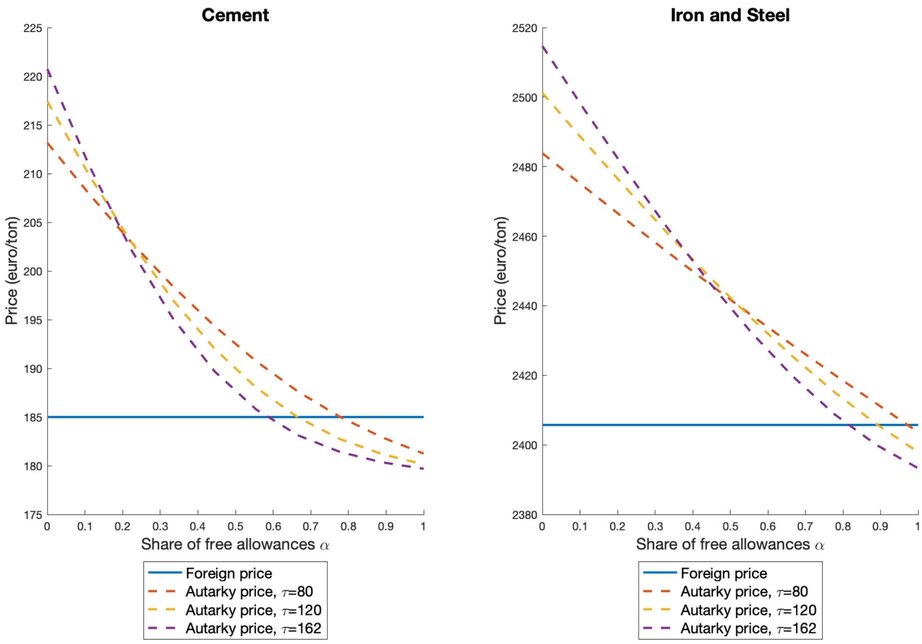

FIGURE 1.9: Equilibrium prices with CBAM.

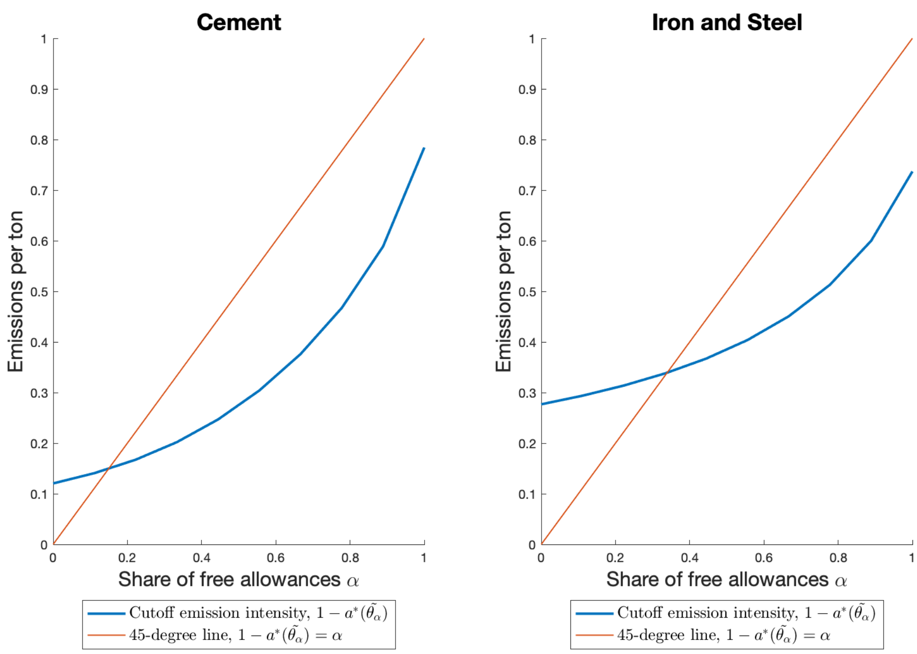

FIGURE 1.10: Cutoff emission intensity, $\tau = 162$.

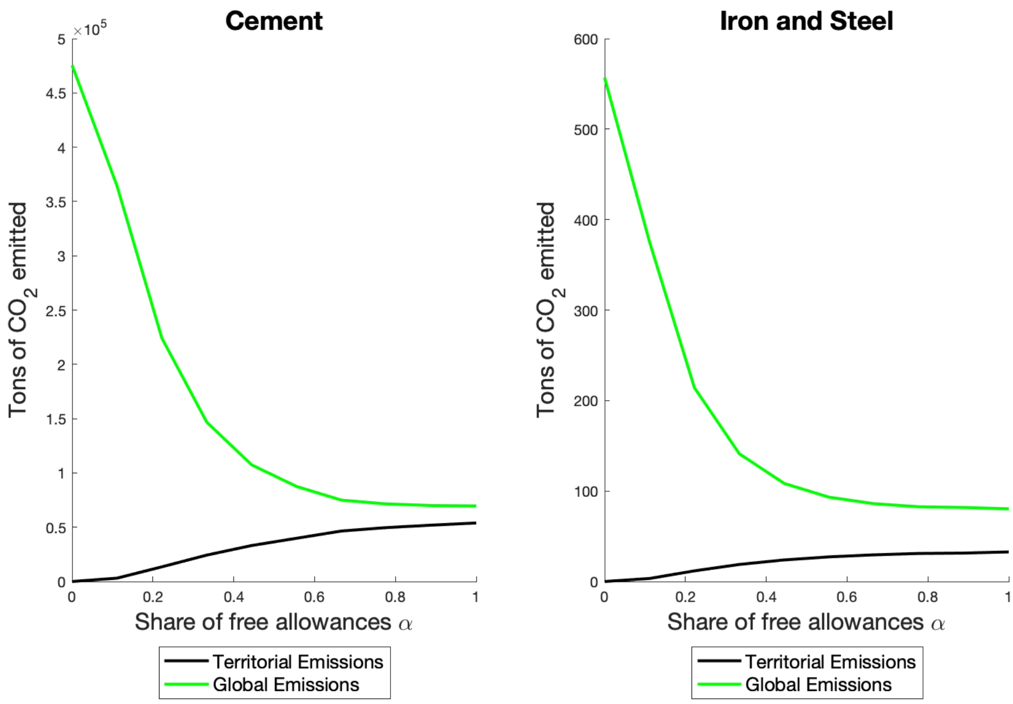

FIGURE 1.11: Emissions without CBAM, with $\tau = 162$.

Robustness of the quantitative analysis

In this section, we gauge the robustness of our quantitative results with respect to the calibrated parameters. First, we estimate β using different years. Using the Italian plant-level data for years other than 2019, we find $\hat{\beta} = 1.48$ for 2018 (using the average τ of 15 observed in that year), and $\hat{\beta} = 0.77$ for 2017 (using the average τ of 5).[37] In Figure 1.12 we plot the welfare under CBAM in each sector (as in Figure 1.7) using the β estimated in different years.

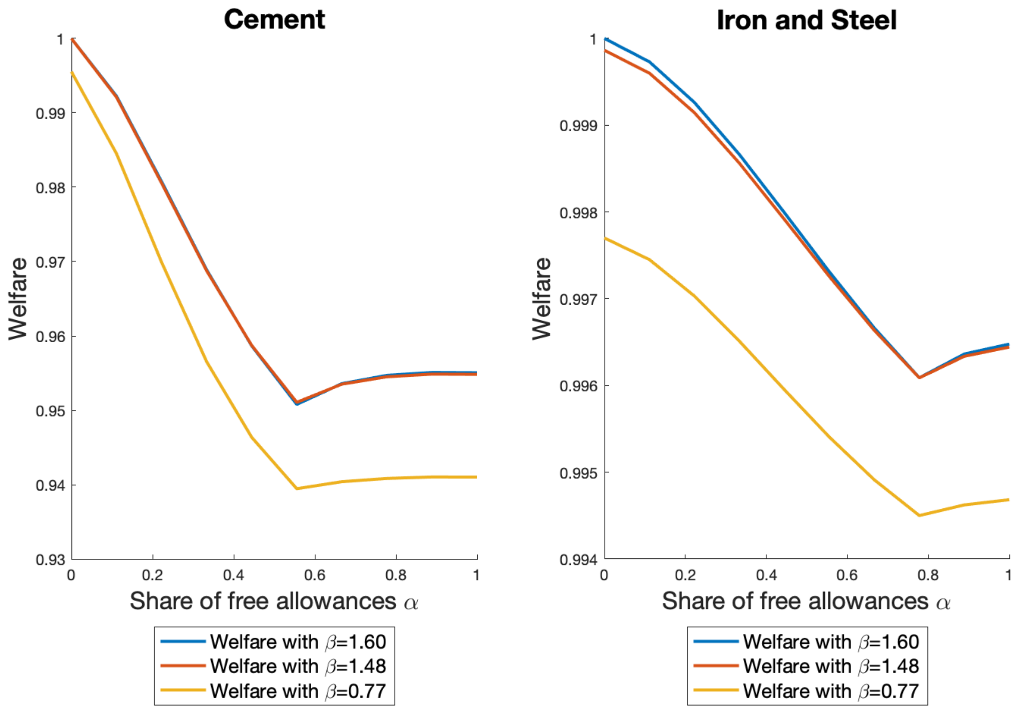

FIGURE 1.12: Welfare with CBAM with β calibrated in different years.

The graph shows that the welfare is close to the baseline welfare for any level for α, but is increasing in the convexity parameter β. Intuitively, when the cost function is more convex, the abatement costs are on average higher, thus the welfare gain arising from the CBAM "protecting" domestic producers from foreign competition becomes larger.

Second, we evaluate the robustness of our results with respect to the values for domestic and foreign emissions. We set $\gamma = 1$, which means that the foreign emission factors are equal to the domestic ones, before abatement.

In Figure 1.13, we can see that the welfare under CBAM is essentially the same as in the baseline for cement, while it is a bit higher for the steel sector. This is because in the steel sector

[37] If instead we use the same τ as in 2019, we find $\hat{\beta} = 1.67$ in 2018 and $\hat{\beta} = 1.09$ in 2017.

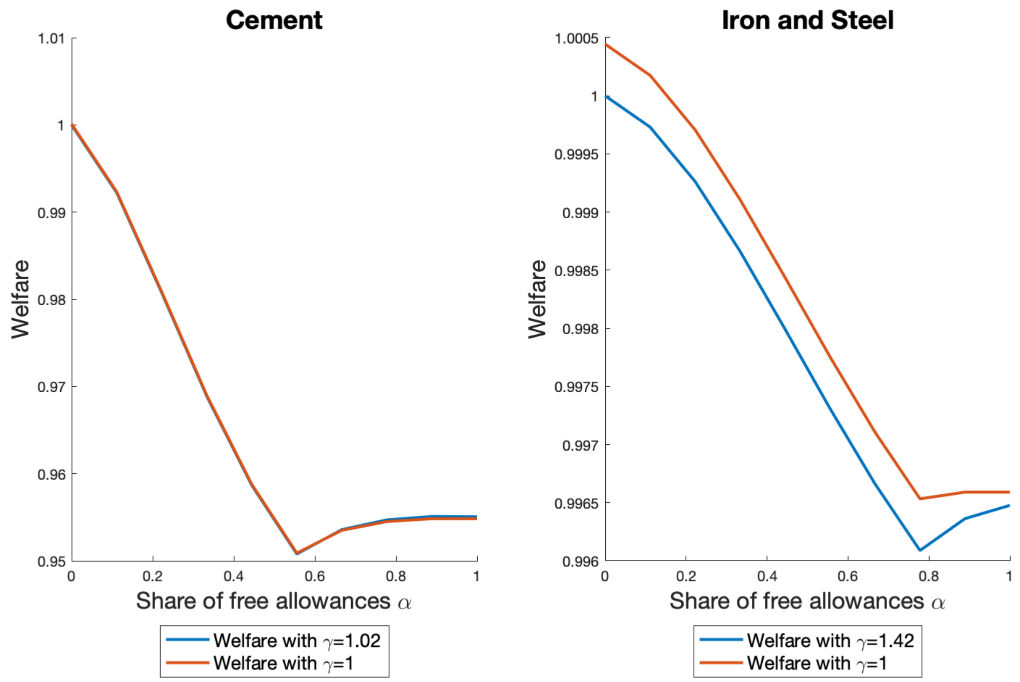

FIGURE 1.13: Welfare with CBAM with $\gamma = 1$.

global emissions are significantly lower than in the baseline, as γ goes from 1.42 to 1. Overall, these robustness exercises indicate that our welfare results are not driven, neither qualitatively nor quantitatively, by the specific point estimates that we impose in our baseline calibration.

Chapter 2

Emissions Abatement and the EU ETS: Testing the Porter Hypothesis

Antonia Pacelli[1]

Do stricter environmental regulations encourage innovation? For answering to this question, this study explores the Porter hypothesis using the Italian data on industrial plant covered by the European Emission Trading Scheme (EU ETS). The heterogeneous shock on free allowances provision in the beginning of the third phase is an instrument to assess the differential impact of this cut on various industrial sectors. This work examines the consequences of reduced free allowances for certain sectors in contrast to those that maintain their existing allocation. By using an indicator of emission intensity based on quantity, this study shows that the presence of free allowances foster the entry of dirtier producers in the market, in line with the "reverse Porter hypothesis".

Keywords: Innovation, carbon pricing, carbon leakage, emission intensity.
JEL codes: D22, H23, L52, O31, Q54, Q58.

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2.1 Introduction

Environmental regulations and policy instruments aiming at fighting climate change and related consequences are flourishing worldwide and becoming stringent and stringent (Eskander, Fankhauser, and Setzer, 2021). In the 20 years after the Kyoto Protocol, the laws on climate change has increased doubling its rate every four to five years (Nachmany et al., 2017). The global environmental crisis and the urgency to deal with climate change consequences are the intuitive reasons that explain this exponential trend in environmental laws. Greenhouse gases and pollution are causing not only the degradation and modification of the natural environment, but also they significantly contribute to the increase of the health risks and diseases' rate (Kjellstrom et al., 2006).

On one hand, concerns related to the cost-effectiveness of environmental policies are crucial from the perspective of policymakers. This is particularly pertinent for domestic and unilateral policy instruments, which embody potential negative effects on competitiveness and productivity of domestic firms with respect to foreigner (and unregulated or less regulated) competitors. On the other hand, theoretical studies explore the benefits for the firms and regulated agents that could arise from environmental climate policies. A leading example is the Porter hypothesis (Porter, 1996; Porter and Linde, 1995), which clarifies that well-designed environmental policies targeted on firms can be beneficial for competitiveness through the innovation channel.

In this work, I would like to empirically explore the Porter hypothesis in the European Emission Trading Scheme (EU ETS) framework. The EU ETS is a marked based regulatory instrument that aims to reduce greenhouses gases (GHGs) emissions, main drivers of climate change. The EU ETS is the first and largest emissions cap and trade system which works on global pollutants. Introduced in 2005, it covers 31 countries (27 EU countries plus United Kingdom, Iceland, Liechtenstein and Norway). It caps the total volume of GHGs from more than 10,000 power stations, industrial plants and airline companies between airports of participating countries, responsible together for around 50% of European GHGs emissions and 5% of global emissions (Ispra Report, 2019). Nowadays, the EU ETS is a milestone of the European climate policy and it represents a leading and virtuous example for many other countries in the world.

The EU ETS covers the direct emissions of firms in carbon dioxide equivalent ($C0_2$eq).[2] Every year EU ETS firms have to surround emission permits (EUAs) that correspond to the quantity of $C0_2eq$ produced. The criteria under which sectors and production plants are covered by the policy are illustrated in the Annex I of the Directive 87/2003/CE. We are currently in the fourth phase of the EU ETS. In the Phase I (pilot phase, 2005-2007) and Phase II (2008-2012), almost all the EUAs were allocated for free, based on historic emissions (grandfathering). Providing allowances for free to some firms which are in the ETS or about to enter in the system

[2]Each allowance gives to the holder the right to emit a ton of CO_2 or the equivalent amount of other powerful GHG, nitrous oxide (N_2O) and perfluorocarbons (PFCs).

is the result of the analysis on carbon leakage, competitiveness and international trade considerations.[3] The free allocation of allowances deals with the risk of carbon leakage by reducing the costs of compliance faced by the operators covered by the EU ETS. Moreover, policy makers justify the existence of the free allowances to safeguard the competitiveness of industries covered by the EU ETS.

The penalty imposed on the companies for non-compliance in the first phase was 40 euros per tonne of CO_2eq. As it is well known, too many EUAs had been freely allocated to plants, and it lead to an oversupply of EUAs and a consequent fall in their price, eventually to zero at the end of the first phase. From 2008 to 2012 (Phase II), the EU imposed a tighter emission cap by reducing the total volume of EUAs by 6.5% compared to the first phase. In 2008 Iceland, Norway and Liechtenstein joined the EU ETS. Within this phase, flights within the borders of the EU ETS countries were included. Up to 10% of the allowances could be auctioned. The penalty for non-compliance became of 100 euros per ton of CO_2eq. The price felt down again from 30 euros to less than 7 euros due to the great financial crisis. 2013 is the beginning of the Phase III, which is the focal point of this paper. The European authorities revised the EU ETS from the third phase in many aspects (European Parliament, 2009).[4] First, it includes an emission cap uniformly over the EU, decreasing by 1.74% per year, to achieve the GHGs reduction target more effectively. More importantly, the main allocation method was modified from grandfathering to auctioning.[5] In 2013, allowances for more than 40% of all verified emissions were auctioned. Based on some benchmarks and carbon leakage considerations, some EU ETS sectors kept the same share of free allowances (as steel, ceramics and refinery) of the previous phase. However, the main reason that made this amendment necessary was the ineffectiveness of the EU ETS due principally to the very low price; moreover, it was subject to several frauds as mentioned in the work Nield, 2011.

The most hoped-for effect of introducing a market of permits to emit CO_2eq is to provide incentives to companies to produce the same quantity with a lower environmental impact. If the policy is well-designed, it potentially can lead to innovation and emission abatement. In order to explore this, the paper evaluates an aspect of fundamental importance for the exploration of the environmental policy outcomes: the relationship between emissions abatement and the reduction of free allowances. First, exploiting a unique dataset[6] covering emissions and quantity produced of Italian plants from 2008 to 2019, for 8 manufacturing sectors,[7] it is built a powerful emission intensity indicator based on quantity produced. This indicator will be considered as a proxy of innovation: for the same amount of quantity produced, if the emission

[3] Carbon leakage refers to the situation that may occur if businesses transfer their production (or, import carbon-intensive intermediate goods) to other countries with laxer emission constraints or where emissions are completely unregulated. This could lead to an increase in global emissions and therefore the scope of the policy might be completely undermined.

[4] Directive 2009/29/EC.

[5] Hepburn et al., 2011 study the different macro effects of auctioning vs free allowances allocation methods.

[6] The Italian dataset through the Convention ISPRA-DiSES. The dataset contains plant-level detailed data on emissions, quantity and plant location, which is relevant for building the emission intensity indicator

[7] agrifood, ceramics, glass, paper and wood, refinery, steel, textile and thermoelectric.

intensity indicator goes down means that the plant, sector or group of sectors has innovated. Secondly, by using the newly build indicator, the study investigates what is the impact of a shock in provision of free allowances on abatement behaviour of firms. This question is particularly attractive since during the Phase III we observe a severe cut in free allowances issuance (in 2013). Performing an evaluation of the impacts at sectoral level, it is possible to capture the heterogeneity of each industrial sector in their willingness to innovate in response to the policy shock. This research contributes to the existing literature by providing insights into the effects and the effectiveness of policy changes within the EU ETS framework. Understanding these dynamics is crucial for policymakers and stakeholders involved in climate mitigation efforts.

The empirical exercise proposed in this research focuses on Italian firms. Italy generates 11.4% of EU's total greenhouse gases emissions, and almost the 10% of the ETS plants are based in Italy. Since 2005, it has reduced emissions at a faster rate compared to the EU average, as reported in the European Parliament Briefing, 2021.

Given the importance of pricing GHGs to reduce emissions, the branch of the literature on the impact of EU ETS has grown fast in recent years. Previous papers have investigated how the EU ETS affects various economic and environmental aspects, including emissions, economic performance, and investments. These studies typically concentrate on individual European countries due to the existence of national registries for detailed firm private data. Wagner et al., 2014 and Colmer et al., 2023 are papers focused on french manufacturing firms, and similar research has been carried out for Germany (Gerster and Lamp, 2023; Löschel, Lutz, and Managi, 2019; Petrick and Wagner, 2014), Norway (Klemetsen, Rosendahl, and Jakobsen, 2020) and group of countries as shown in Dechezleprêtre, Nachtigall, and Venmans, 2023, which includes the United Kingdom, France, Norway, and the Netherlands. These studies consistently identify a reduction in carbon emissions of approximately 8-10% in the first phases of the EU ETS (up to 2012), without observing significant effects on economic performance indicators such as employment, value added, carbon leakage, and/or investments. Naegele and Zaklan, 2019 reviews the studies on carbon leakage and the authors so far do not find any evidence of carbon leakage caused by the introduction of EU ETS. This studies focus on the first two phases of the policy (2005-2012), in which prices were pretty low and the allowances for free were over allocated. Moreover, given the low cost of the emission permits, it becomes challenging to rationalize decisions such as relocating production facilities or shifting supply chains towards goods imported from unregulated regions. Compared to the literature, this paper extends the analysis to the third phase of the EU ETS. Beyond simply extending the temporal scope, this phase exhibits a more dynamic and increasing price and nd pivotal revisions pertaining to permit allocation rules. It is important to note that many of these papers acknowledge the potential role of free allowances within the EU ETS, although they generally do not conduct extensive investigations into their impact on the outcomes under consideration. Specifically, our study zooms in on the effectiveness and impacts of the regulatory instruments embedded within the framework. We analyse the strategic implications of subsidizing production plants through the provision of free allowances, which could potentially impact the abatement decisions of firms, contingent upon factors such as permit pricing, abatement costs, and opportunity costs. To the best of our knowledge, this paper pioneers an empirical study to empirically capture the impact of the presence or absence of this policy tool.

Related to the present paper, a feature discussed in the literature is how to define emission intensity. Generally, it is defined as the ratio between emissions and GDP (European Parliament Briefing, 2021), or emissions and value added, revenues or employment variables (Calel, 2020; Colmer et al., 2023; Klemetsen, Rosendahl, and Jakobsen, 2020; Wagner et al., 2014), or gross output (Petrick and Wagner, 2014). Klemetsen, Rosendahl, and Jakobsen, 2020 is the only that uses a measure of emission intensity related to the electricity consumption. A recent working paper by Belloc and Valentini, 2023 derives the emission intensity using the allocation rule to estimate the quantity produced. However, being based on median values, the measure is not exempt to measurement errors and misalignment. By using the rich dataset at plant level of the Italian Registry, the contribution of this work is to construct a pure indicator of emission intensity based on quantity. The primary advantage of this pure indicator lies in its resilience against economic perturbations, which potentially alter value added, employment and economic performance metrics.

Related to the investment branch of the literature, some papers focus on the potential effect on innovation (Calel, 2020; D'Arcangelo and Galeotti, 2022) and productivity (D'Arcangelo, Pavan, and Calligaris, 2022). Calel and Dechezleprêtre, 2016 find that the EU ETS has increased low-carbon innovation among regulated firms by around 10%. The present paper adds an additional feature to this result: the implementation of low-carbon technologies mainly comes from the new entrant firms in the EU ETS, belonging to the sectors that faced a cut in the free allowances provision.

The main finding of this work is that the cut of free allowances has an impact on the emission intensity at sectoral level. In particular, the group of sectors facing a reduction in free allowances experiences a decline in emission intensity compared to the average plant of the others that continue to receive full or same share of subsidies, that instead becomes "browner" in terms of emission intensity. So, when the policy becomes more stringent, in the treated sectors the average plant becomes more innovative and efficient, producing in a less impacting way on the environment; with a lenient policy frame, there are no incentives to clean the production. More in details, this study shows that the result is mainly driven by new entrants, at least in the short-run. The absence of free allowances does not provide direct incentives to abate emissions, but the reduction fosters the entry of more sustainable producers. It is consistent with the assumption made in the theoretical model of Ambec, Esposito, and Pacelli, 2023. Analysing the side of subsidized sectors, it is even clearer in our study that for them the presence of free allowances fosters the entry of dirtier producers. This finding is in line with the "reverse Porter hypothesis", which changes the perspective of the Porter's idea: with a lenient or lacking policy does not provide incentives to explore the innovation channel as beneficial for competitiveness and economic outcomes of the firm. Moreover, emission intensity is measured as an estimate of the direct CO_2eq embodied in each product, representing a novel approach in the literature. This lack in the previous studies is mostly due to the limited availability of plant-level production data.

Can institutional authorities learn from the Italian case? This empirical evidence that the Porter hypothesis holds for a more stringent design of the EU ETS, relying on less free allowances and lead to cleaner plants in the not subsidized sectors. Focusing on the role of free allowances as policy tool, the contribution of the paper to the existent literature is the analysis of the effect of reducing free allowances on the emission intensity. Since the aim of the policy is to effectively and efficiently reduce the emissions (at least cost for firms),[8] it is crucial to evaluate the optimal policy tools mix to achieve these objectives.

The paper proceeds as follows: first, Section 2.2 focuses on the Porter hypothesis and the channels explored in the EU ETS context. Then, it describes the context and state of the art of the EU ETS in Section 2.3. In Section 2.4 are presented the data and the methodology, followed by Section 4.6 which illustrates the results. The summary and the conclusions are in Section 2.6.

2.2 The Porter Hypothesis

The effects of environmental regulation on innovation has an extensive economic empirical and theoretical literature, but with contrasting results (Ambec et al., 2013; Yu and Zhang, 2022).

The Porter hypothesis formulated and clarified in the works of Porter and van der Linder (Porter, 1996; Porter and Linde, 1995) states that firms covered by an environmental regulation can potential gain private benefits through the innovation channel, and the consequent enhancement of productivity. In this view, climate policies are not only beneficial for the environmental outcomes and socially desirable, but are also politically supportable for the benefits gained by agents populating the economic fabric of the interested region. In contrast, the other side of the literature argues that stringent environmental policies incorporates costs for the firms and affect the innovation budget (Gray and Shadbegian, 1998; Greenstone, List, and Syverson, 2012).

More in details, the empirical and theoretical literature based on the Porter hypothesis analyses on the relationship between environmental regulation and firms' competitiveness, passing through the innovation channel (Brännlund, Lundgren, et al., 2009). The empirical literature does not generally support the Porter hypothesis, since it does not find a so called "Porter effect", or even negative effects on economic outcomes (Gollop and Roberts, 1983; Gray and

[8] Article 1 of the European Parliament, 2003 Directive 2003/87 clarifies that the aim is "the reduction of greenhouse gas emissions in a cost-effective and economically efficient manner". The presence of free allowances might influence and drive emission abatement decisions.

Shadbegian, 1998). However, an interesting study of Jaffe and Palmer, 1997 finds a positive effect of environmental regulations on research and development investments.

Among the theoretical studies, Ambec and Barla, 2002 provides a formalization of the Porter hypothesis, in particular on how the introduction of a regulation can lead firms to innovate. In their model, the authors show that environmental regulations could increase investments in research and development and firms' profits, given that less polluting technologies are more productive.

The idea in this paper is to use an alternative proxy of the innovation, in order to strictly focus on the channel of emission and efficiency: the emission intensity based on the output produced. In the context of the EU ETS in Italy, using the free allowances shock of the third phase, it is possible to analyse the differences among plants and sectors that need or do not need to pay the permit price. Moreover, this study changes the perspective of the Porter hypothesis: does a lenient policy context foster the entrance in the market of dirtier producers? Hence, what Porter suggests is that environmental regulations fosters innovation, but is it true as well that the absence of environmental innovation helps no-innovating and dirty producers to profitably entry in the market? This is what here it is called the "reverse Porter hypothesis".

2.3 Free Allowances and Emissions

The practical effectiveness of the EU ETS relies on plants paying the permit price for their emissions. The allocation of a significant share or the provision of total free allowances (≥ 80) can potentially influence the entry and exit of environmentally unsustainable producers and affect emission reductions at the sectoral level.

The theoretical model presented by Ambec, Esposito, and Pacelli, 2023 shows that the presence of a high share of free allowances does not provide incentives to abate emissions since "offsets" the policy, but also it fosters the entry of carbon intensive producers. This idea shows from another perspective part of the reasoning behind the Porter hypothesis (reverse Porter hypothesis), where an absent or more lenient environmental policy does not provide incentive towards innovation and, on the contrary, allows dirty producers to be on the market and being profitable. Does it describe the mechanisms in the real world?

Figure 2.1 reveals a substantial reduction in the issuance of free allowances in the first year of Phase III (2013-2020), due to the implementation of the European Parliament, 2009 Directive 2009/29/EC. The figure illustrates also the annual $C0_2eq$ emissions tonnes in Italy from manufacturing plants covered by the EU ETS policy. The emissions exhibit a declining trend, which intuitively aligns with the objectives of the policy to reduce GHG emissions. However, as previously mentioned, the underlying aim of this European directive is to achieve emission reductions without compromising the economic performances of the regulated firms. Therefore, while a decreasing emissions trend appears favorable, it is crucial to examine whether it is

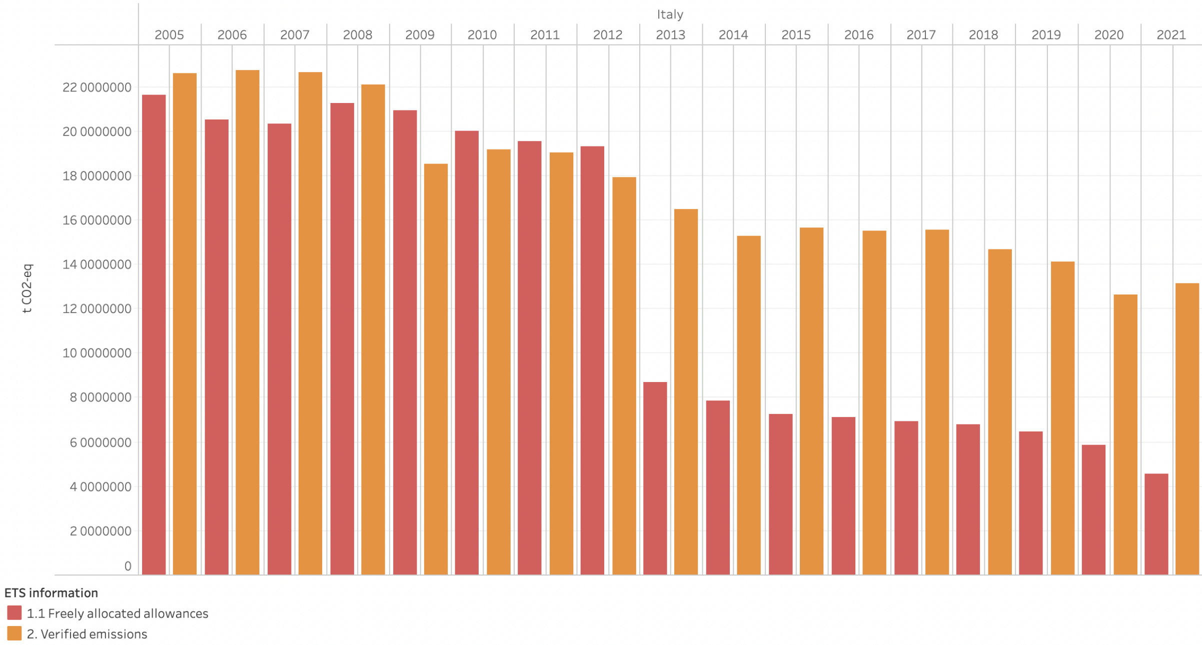

FIGURE 2.1: Free allowances and verified ETS emissions in Italy, all sectors. Source: European Environmental Agency data viewer.

driven by a shift towards low-carbon production or due to underproduction. This aspect holds significant relevance in evaluating the intended and unintended effects of the policy.

2.4 Data and Methodology

The objective of this section is to provide a description of the dataset used with a particular focus on the components of principal metric driving the analysis: the emission intensity. It is essential to analyse its two components (quantity and emissions) to comprehend the economic implications of the policy and its instruments. Following the illustration of the data, the section proceeds to show the methodology employed in the data analysis.

2.4.1 Data

The database utilized for this study incorporates data from multiple sources, including the publicly available EUTL (European Union Transaction Log) platform, the European Environmental Agency (EEA) data source, and the Italian Registry of the EU ETS.[9] The data goes from the EU ETS implementation (2005) to the year before the COVID-19 pandemic (2019). However, due to the data-lacking in the Registry for the Phase I, the analysis is focused on the Phase II and III (2008-2019). The analysis includes eight sectors, considering both 2-digit and 4-digit NACE classifications, as well as specific product-level analyses. The sectors and corresponding NACE codes are outlined in Table 2.2 in the Appendix 2.6.

[9] Access to this dataset was obtained through the ISPRA-DiSES Convention (2021-2024), ensuring compliance with privacy rules. The data provided are aggregated at the sectoral level, preserving confidentiality.

The inclusion of sector-specific considerations arises from the heterogeneity in the 2013 cut of free allowances across sectors. The reduction in free allowances predominantly affected sectors such as thermoelectric, agrifood, textile, glass, and paper. In contrast, the remaining sectors either retain full subsidies or continue, as in the case of refinery sector, receiving an equivalent proportion of free allowances. Hence, the main variables in the analyses are plant-level observations of emissions, quantity produced and allocated free allowances.

It is essential to underscore that the sample utilized for this analysis exhibits an unbalanced nature, attributed to the dynamics of the EU ETS framework. Indeed, this unbalanced nature arises from plants opting in and out the EU ETS (see Figure 2.2). [10]

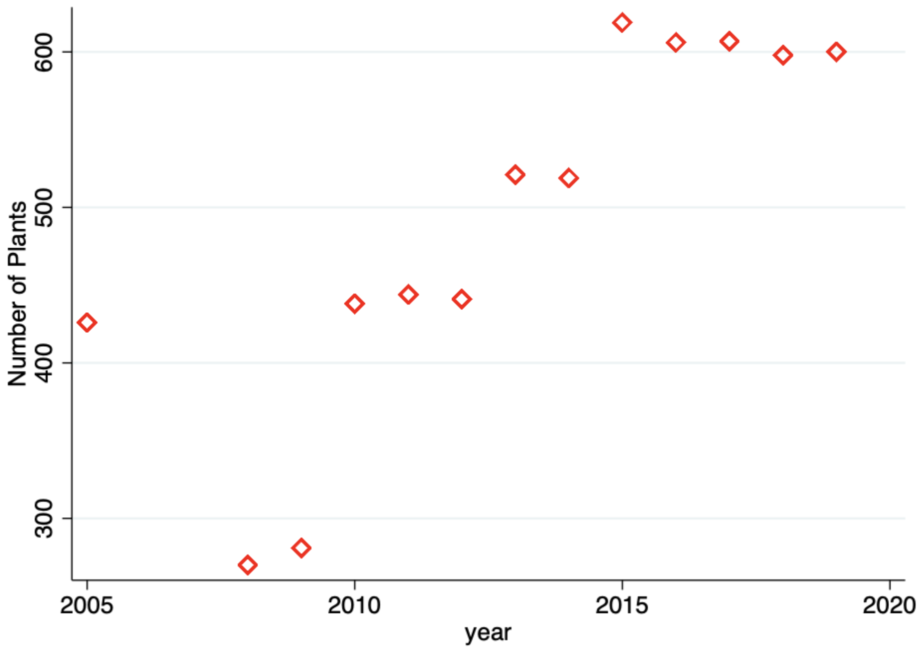

FIGURE 2.2: Number of plants in the sample.

The transition from Phase II to Phase III of the EU ETS witnesses a substantial number of plants undertaking the opt-in and opt-out options. Nevertheless, the unbalanced nature of the panel does not affect the results of the main analysis, since we work on the emission intensity indicator, and not on the absolute value of the emissions.

Quantity and emissions

The emission intensity metric has two components: emissions and quantity. The first stage involves comprehending the temporal trajectory of these components and elucidating their interrelationship. This endeavor aims to discern the distinct impact of an incremental unit of quantity produced on emissions by each plant. This analysis helps to disentangle fixed and

[10] In the EU ETS, it can happen that firms opt out by the system. The reasons are twofold: first, the plant/firm can close its activity; second, the production plants change technology and go under the EU ETS threshold. For the Italian plants, they continue to declare their emissions in the registry of small emitters called RENAPE.

variable environmental costs. Fixed environmental costs are independent from the amount of quantity produced - for instance, machinery that constantly operates. In contrast, variable environmental costs exhibit a direct correlation with quantity, and any fluctuations in the latter generates alterations in the former.

Overall, the quantity produced by EU ETS plants has increased over time in the sectors considered in this analysis, as Figure 2.3 shows. One reason of this increasing trend depends on the increased number of plants over time (from 400 to 600 plants in the Italian sample). However, also considering only the plants that have observations before and after 2013, the quantity has an increasing trend.

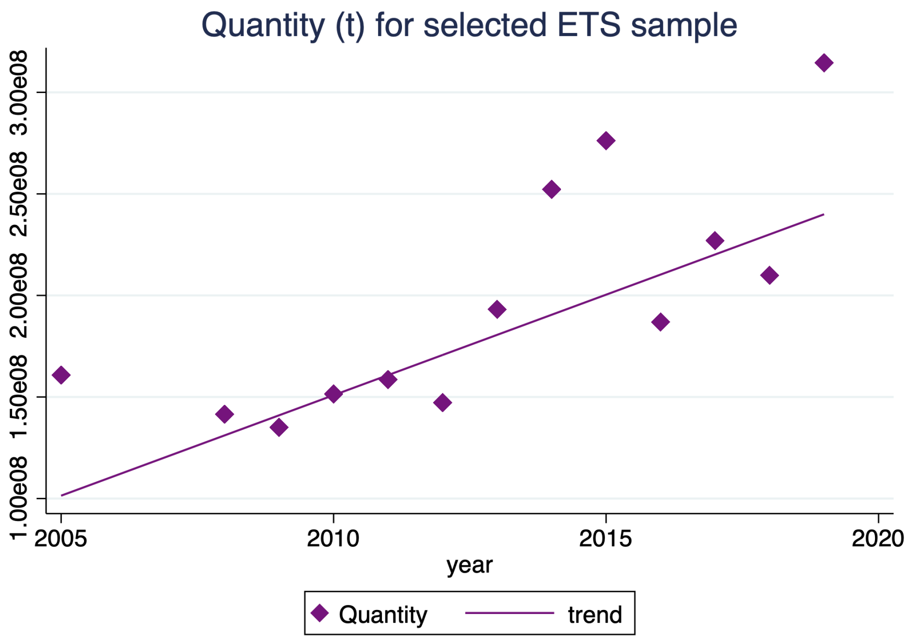

FIGURE 2.3: Industrial production for selected Italian plants, excluding thermoelectric.

Figure 2.4 provides a flavour of the heterogeneity across sectors in the quantity trends. It is possible to generally observe an increasing trend, except for glass and thermoelectric sectors, where it is pretty stable or decreasing. In the glass sector in 2013 an 2014 some firms misreported the data: production data are not on tonnes but number of pieces or square meters, which makes hard the conversion in tonnes. So, the trend in the production of glass is stable and does not experience any drop in those years. Without considering new entrants and opting out plants, sectors as agrifood, textile and refinery have an increasing trend in production, whereas in the other sectors we observe a stable or decreasing trend. Ceramics is a sector that experienced a relevant opted in in 2013. [11]

The other component of interest is the level of emissions. Figure 2.5 exploits the path of emissions, considering exclusively the plants situated within Italian borders across the specified

[11] However, the plants already covered before the beginning of Phase III overall reduced their production.

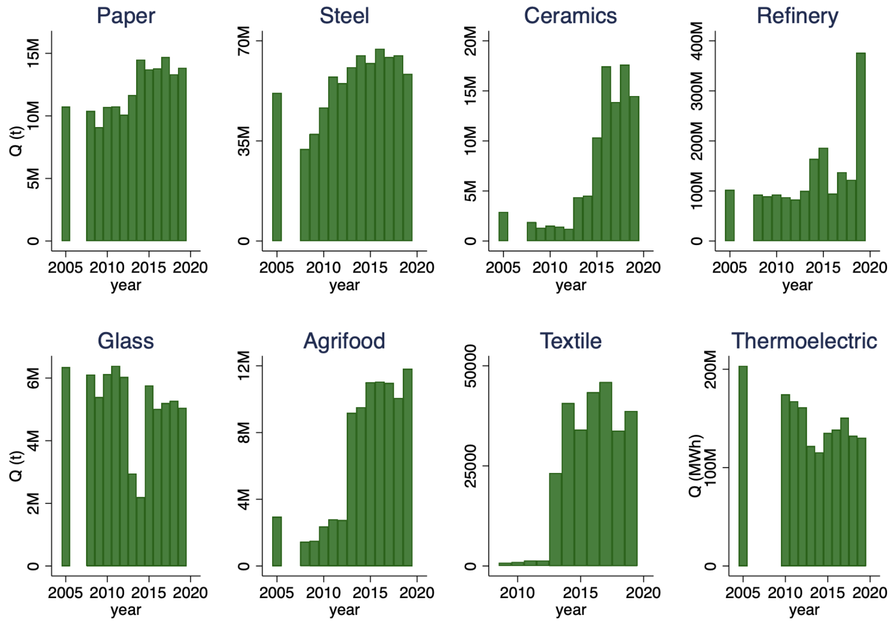

FIGURE 2.4: Quantity produced for each selected Italian sector.

sectors (sourced from EEA data). The graph distinctly illustrates a decline in emissions commencing from 2009, coinciding with the period of the Financial Crisis. Concurrently, it has been already discussed that the year 2009 marks the announcement of reforms for the EU ETS in preparation for Phase III (2013). Over the observed span, the emissions path exhibits a pervasive downward trend, signifying a positive signal for the EU ETS. However, it is important to note that while this downward trend is encouraging, it alone does not suffice to conclusively affirm the efficacy of this policy or to assert that it has not engendered adverse consequences on the operational efficacy of the encompassed firms.

This pivotal consideration forms the foundation for the initial emphasis on the quantity dimension in this work. The computation of the emission intensity indicator, which gauges emissions in relation to production volumes, serves as a strategic approach. By evaluating the metric of emissions per unit of production, a more sophisticated assessment emerges. This approach enables the research to delve beyond the overarching emissions trend and consider how and why emissions reduced.

To precisely assess whether the observed trend reflects a transition to low-carbon production or underproduction, this paper builds the measurement of emission intensity. This measure is a valuable contribution to the literature: it is a novelty because based on quantity produced, data not generally available.

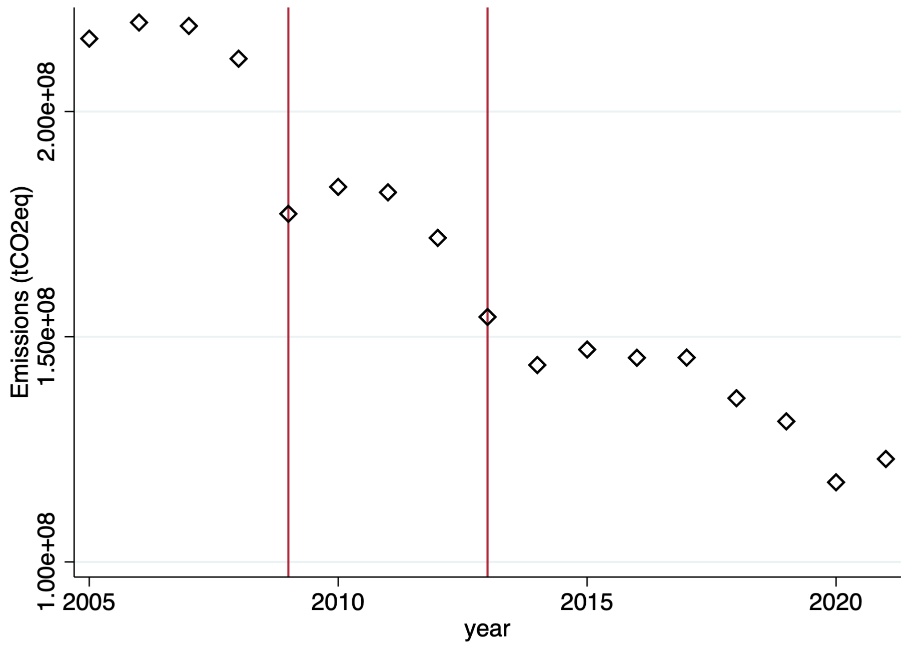

FIGURE 2.5: Path of Verified Emissions for the selected Italian sectors.

2.4.2 Methodology

Exploiting the unique features and richness of this dataset, the present paper creates the emission intensity (EI) indicator, computed as the ratio between emissions (e) and quantity (q) produced annually, both measured in tonnes. The only exception is for thermoelectric sector, where the output is measured in megawatt per hour (MWh).

The emission intensity computation is described from the following equation:

$$EI_{i,t} = e_{i,t}/q_{i,t} \tag{2.1}$$

The EI indicator is therefore based on two components: emissions and quantity produced. The path of EI provides an important piece of information: considering an increasing or stable path in production quantities, if the emission intensity decreases over time it means that firms are investing in technologies and innovation for producing the goods in a more sustainable way. In this context, a decreasing emission intensity path is an hoped result for the EU ETS policy, because it represents a good signal of greener production without undermining economic performances and the production levels. On the contrary, an increasing path it is a bad signal for the environmental standards of the production, which is not desirable. Moreover, an increase in the EI is a possible scenario in period of crisis, where fixed environmental costs are constant (e.g., machinery that cannot be turned off) but production goes down.

The emission intensity indicator is used for answering to the main research question addressed in this paper: what is the role of free allowances on abatement behaviour of firms? In other words, the empirical exercise proposed assesses the effects of the reduction in the provision of free allowances on the emission intensity. Figure 2.6 illustrates the share of free allowances per sector over time. Some sectors (specifically paper, thermoelectric, agrifood, textile industries[12] and glass) underwent a notable reduction in their allocation of free permits in 2013. Conversely, several other sectors maintained a substantial or complete share of allowances allocated for free. This observation implies that, despite their formal inclusion within the EU ETS, these sectors do not fully or directly internalize the cost associated with emissions through the permit pricing mechanism. An interesting case is the refinery sector, which encountered a reduction in free allowances at the beginning of Phase II (2008). In 2013, the allocation of free allowances for refinery plants did not experience a shock. Therefore, the sectors allocated in the treated group are the ones that experienced a shock from the Phase II to the Phase III. Conversely, in the control group the sectors did not observe a significant change due to the Phase III amendment.

The contextual circumstances presented provide a conducive environment for employing the difference-in-differences methodology as the chosen analytical approach. The difference-in-differences approach is employed to disentangle the specific effect of interest. The treatment is the shock in free allowances allocation, wherein specific sectors encountered a substantial reduction in their free allowances in 2013, thereby necessitating the procurement of emission permits from the market. In this case, the treated sectors are paper, glass, agrifood and textile. In contrast, other sectors continued to receive substantial allowances, some even obtained them entirely free of charge: steel, refinery and ceramics.

Differences-in-differences regression constitutes a methodological tool utilized to discern the causal impact of an event. This entails a comparative analysis between units that experienced the event (treated) and those that remained unaffected (control). This methodological approach is grounded in the foundational principle that in the hypothetical absence of the event, the discrepancy between the treated and control groups should remain constant over time. This assumption is fulfilled if the trends in the pre-shock period are parallel. In our case, we show that this assumption is satisfied. The baseline model in this paper has the following specification:

$$EmissInt_{i,t} = \beta_0 + \beta_1 did + \beta_2 T + \beta_3 POST + \epsilon_{i,t} \tag{2.2}$$

The dependent variable is the emission intensity (EmissInt), at plant level and at time t. *T* is the dummy indicating whether the plant belongs to the treated or to the control group. *POST* is zero before 2013, and 1 from 2013, when the cut in free allowances occurred. The variable *did* is a dummy variable created interacting *POST* and *T*. To the baseline, in the Model 1 we add sector and time fixed effects to control for sector unobservable heterogeneity.

[12]Thermoelectric, agrifood and textile are considered together as "combustion" for the EEA

$$EmissInt_{i,t} = \beta_1 did + \beta_2 new\#T + \lambda_t + \lambda_s + \epsilon_{i,t} \tag{2.3}$$

Specification of the regression (Model 2) considers the impact of the new entrants, adding the interaction between the dummies *new* and *T*.

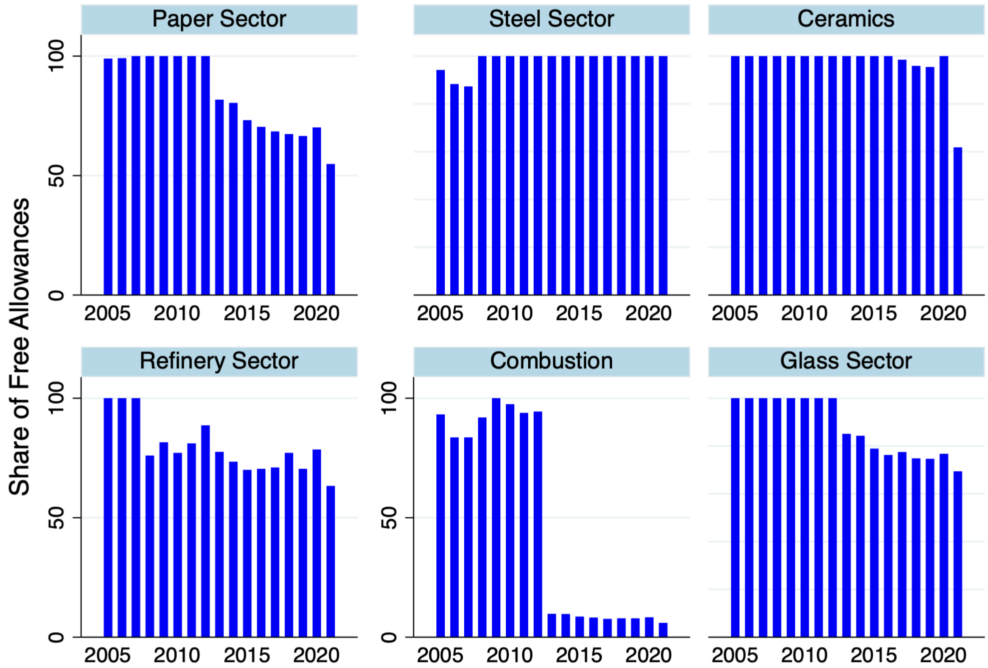

FIGURE 2.6: Share of Free Allowances for selected Italian Sector. Data source: EEA.

The descriptive analysis intuitively suggests that generally the free allowance provision was massive and over-estimated up to 2013, as it is clearly showed in Figure 2.7. The over-estimation comes principally from the difficulties to estimate the allowance needs in advance. The estimates were based on historical emission data, however the collection of environmental data in Italy starts with the announcement of the EU ETS (2000): before this policy, the dataset was scarce and not accurate. Furthermore, it is worth highlighting that the allocation of free allowances also extends to new entrants. This accounts for the observed peak in the ceramic industry, attributed to the inclusion of numerous new ceramic firms in the beginning of the third phase.

The most significant decline in the proportion of free allowances is evident within the glass sector and the broader combustion industries, encompassing agrifood, textile, and thermoelectric sectors.[13]

[13]While the thermoelectric sector primarily drives this reduction, a substantial drop is also verified in the agrifood sector in 2013.

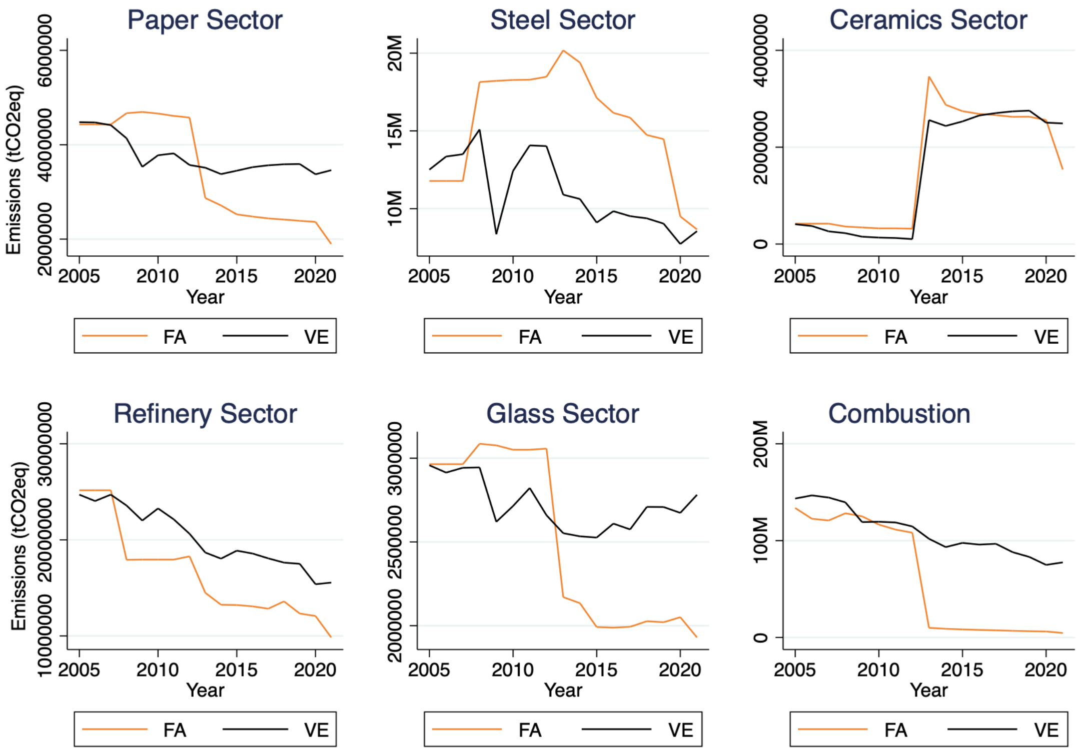

FIGURE 2.7: Free Allowances and Verified Emissions in Italian sectors.

Quantity and emissions in treated and control group

The quantity produced has an increasing trend both for the treatment and control groups. In levels, the quantity produced in the sectors belonging to the control group is higher than the treatment group ones, even if they have less production plants. This disparity in production quantity is attributed to the variations in production volumes, with the control group exhibiting higher volumes compared to the treated sectors.

Figure 2.8 illustrates a positive trend for both groups, signifying an encouraging trajectory. The upward trend observed in the quantity variable serves as a favorable indication for the efficacy of the ETS policy. The thermoelectric sector is excluded due to its distinct output measurement unit (MWh instead of tons).[14]

In the carbon pricing era, a prominent concern revolves around its potential impact on the economic performance of firms. Policymakers advocate for the implementation of such policies under the principle of *least cost*, implying that any reduction in emissions should not undermine production output. Consequently, the observed positive trends depicted in Figure 2.8 intuitively suggest that the implementation of the EU ETS has not adversely affected production levels. This would align with the broader empirical literature on the EU ETS, which commonly does not find a significant impact on the economic performance of firms. Prominent

[14] See Appendix 2.6.

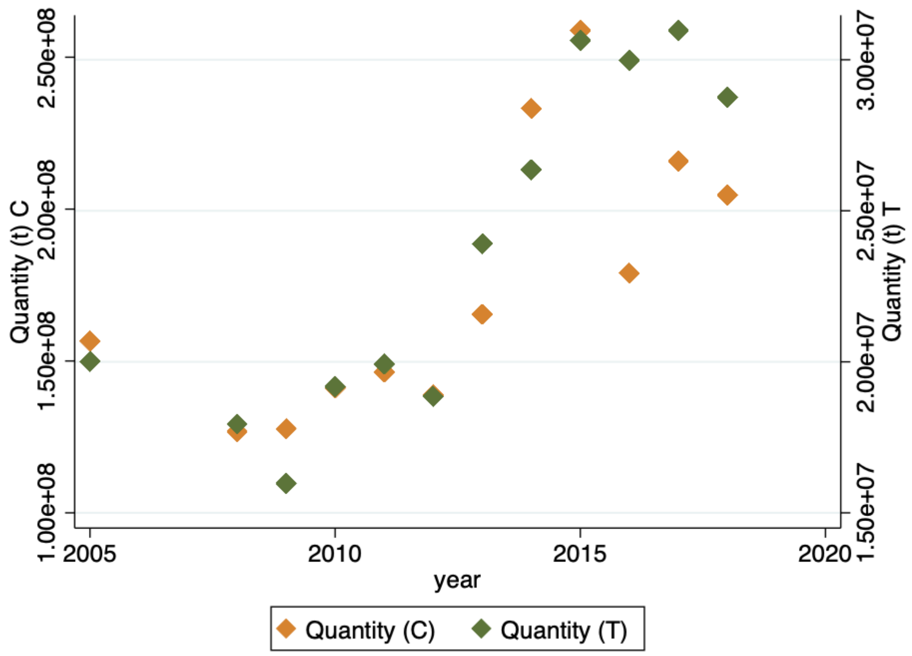

FIGURE 2.8: Quantity for the control and treatment groups, excluding thermoelectric.

papers within this literature include Wagner et al., 2014 and Dechezleprêtre, Nachtigall, and Venmans, 2023.

However, the progressive increase in production volumes depicted in Figure 2.8 is not solely attributed to a surge in production intensity but is also influenced by the escalating number of EU ETS plants over time in the sectors considered. This dynamic becomes clear when considering the evolving count of plants, which displays an upward trajectory over the same period, as depicted in Figure 2.9. This trend is observable across both the treated and control groups.

Analyzing the emissions data, a significant reduction is evident within the control group. Conversely, the treated group (excluding thermoelectric) shows a marginal increase in emissions. However, these observations underscore the motivation underlying the focus of this work on emission intensity at the plant level. While an overview of these emissions data offers valuable insights, it is important to recognize that drawing conclusions about the impact of the EU ETS on abatement choices of plants necessitates a deeper approach. Emission intensity emerges as a pivotal metric, quantifying the ratio of emissions in metric tons to the quantity of product produced over time.

It is noteworthy that the thermoelectric sector exhibits a notable downtrend in both quantities produced and emissions. This trend is partly attributed to the significant number of plants that opted out of the EU ETS in the beginning of the Phase III (see Appendix 2.6 for further details).

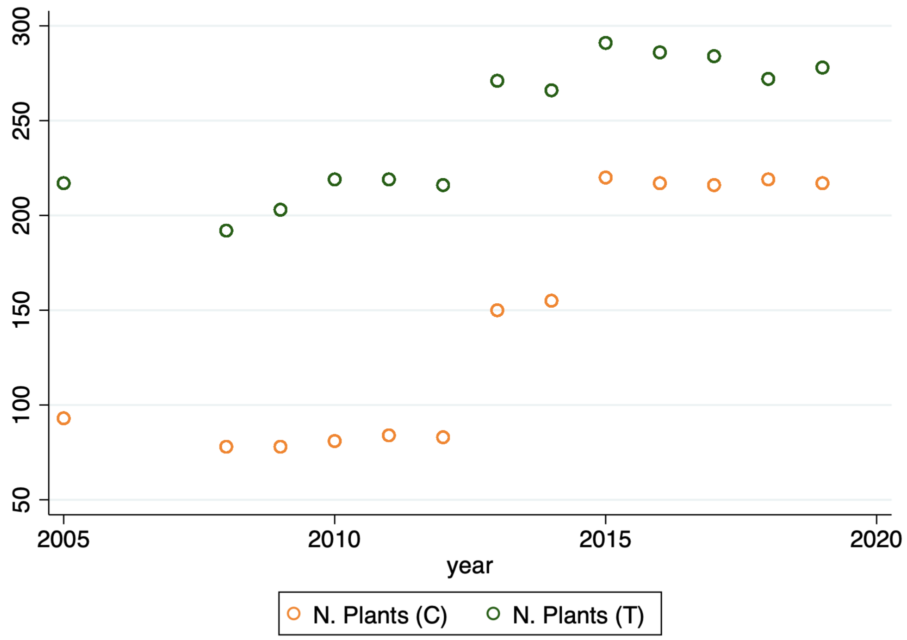

FIGURE 2.9: Number of plants in the treated and control groups, excluding thermoelectric.

Before delving into the findings pertaining to the aforementioned potential influence of the free allowances cut on emission intensity on the whole sample, the subsequent paragraphs elucidate the mechanisms underlying the second part of the research: an exploration of the contributors to emission intensity reduction, specifically examining whether it stems from plants that have been part of the EU ETS since its inception in 2005 or from newly entered participants. In this study, these two distinct groups will be referred to as the "established" group, comprising plants that have been part of the EU ETS since 2005, and the "new entrants", encompassing recently incorporated plants that joined the EU ETS at a later stage (Phase III).

Emission Intensity

This paper has so far presented a descriptive analysis of the temporal trends in quantity and emissions in Italian EU ETS plants over the years 2008-2019. On initial examination, there emerges a perceptible increase in the overall quantity produced by firms operating within the EU ETS framework, coupled with a corresponding decline in emissions. Intuitively, this observable trend suggests a positive effect in terms of the efficacy and outcomes of the EU ETS policy.

However, the core of this analysis delves beyond the interpretation of the output in terms of production tonnes and emissions. The underlying mechanisms driving these trends requires deeper exploration: it is interesting to investigate specific sectors that may lead these shifts. Furthermore, a more detailed analysis seeks to distinguish between two scenarios: the potential emission reductions by firms that have been participants since the inception of the EU ETS, versus new producers entered over time.

Examining the trajectory of emission intensity as illustrated in Figure 2.10, experiences a decreasing pattern. The graph displays a declining trajectory from 2009 until 2014, followed by a subsequent upward trend until 2018. This temporal evolution in emission intensity is indicative of a dynamic and potentially influential phenomenon. The figure that includes thermoelectric plants has a similar path (see the Appendix 2.6).

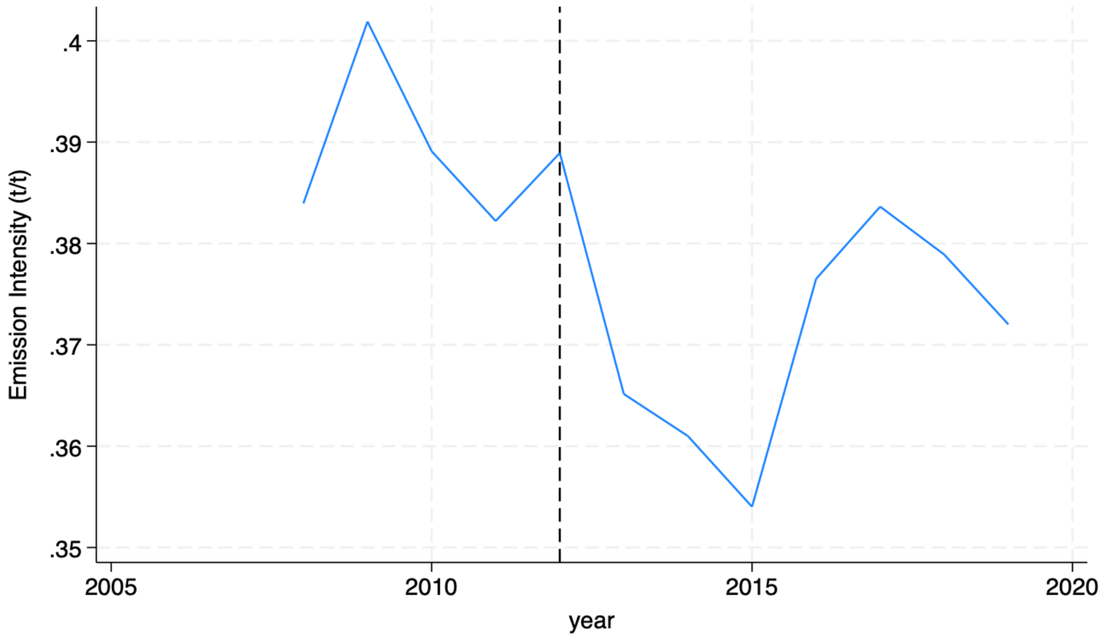

FIGURE 2.10: Path of Emission Intensity for the selected Italian sectors, except thermoelectric.

2.5 Findings

This section presents the findings derived from the comprehensive analysis of the effects of free allowances reduction on emission intensity within the context of the EU ETS. Building on the preceding descriptive exploration, the following paragraphs delve into the core outcomes of this study. Specifically, it aims to unravel the relationship between policy changes and industrial dynamics, shedding light on how these factors collectively shape the emission intensity trends. The analysis begins by examining the alterations in emission intensity across sectors within both the treatment and control groups. This examination serves as a foundational step in understanding the patterns that might indicate policy-induced shifts. Additionally, the influence of new entrants is examined, as it is a factor that can significantly affect the outcomes of the analysis. By dissecting the role of these entrants, this study aims to provide a perspective on how policy changes interact with industrial evolution, highlighting both anticipated and unforeseen consequences. Through these findings, this analysis contributes to a deeper understanding of the impacts of policy interventions, such as the EU ETS.

2.5.1 Impact on the Emission Intensity

Emission intensity is calculated as the ratio of tonnes of CO_2 emitted to tonnes of quantity produced (or MWh for the thermoelectric sector). The cement sector is excluded from the main manufacturing sectors due to the confluence of ETS Phase III changes with the Directive 2010/75 EU and the BAT Directive. Including thermoelectric plants does not significantly alter the graph, as the scale of emission intensity remains consistent. The estimates of emission intensity exhibit a consistent alignment across sectors within both groups. However, since the sectoral composition in the treated and control group slightly changes over time, it is reported the emission intensity by sector in Figure 2.11.

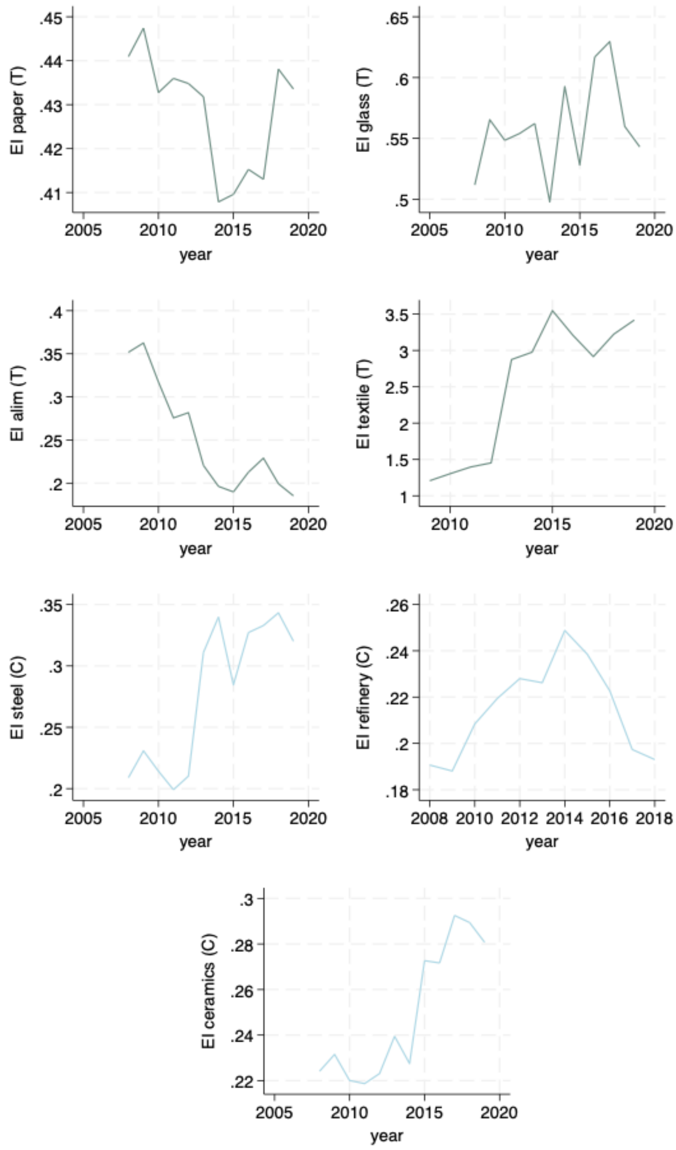

FIGURE 2.11: Emission Intensity path in different sectors.

In the treated group of sectors, with exception of the agrifood sector, the emission intensity path is overall stable or increasing. Therefore, for these sectors the innovation and technological improvement channels do not seem to be explored by the plants in the sample. Therefore, the Porter hypothesis does not hold in general and the policy has not reached the desired target for being considered a "well-designed" policy in Porter terms so far. However, what instead happens in the control group is that the emission intensity is increasing for steel and ceramics sectors.

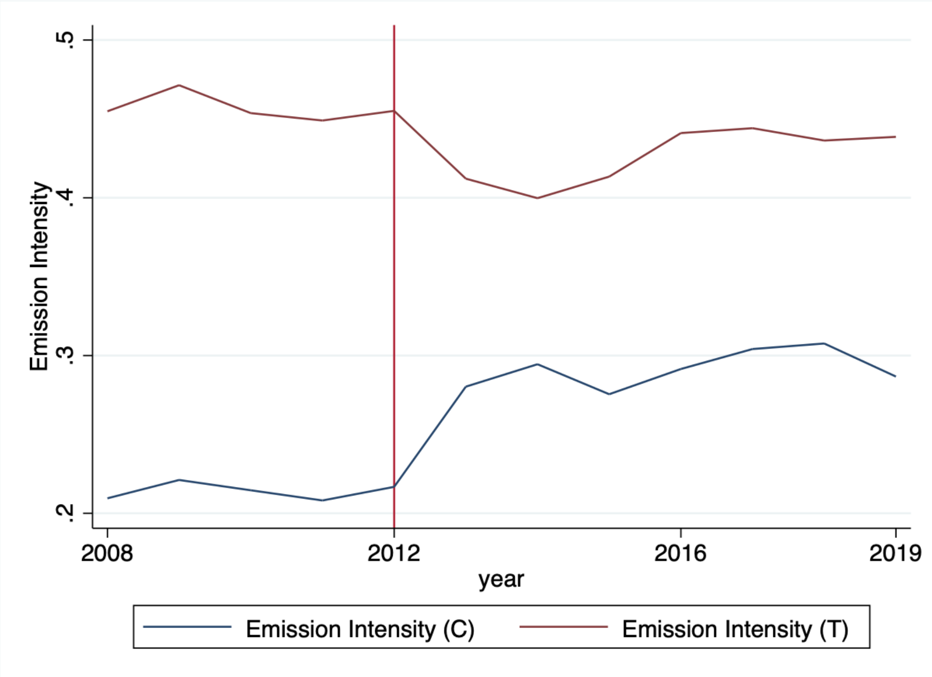

FIGURE 2.12: Emission Intensity, treated and control group (excluding thermo-electric).

The result described in Figure 2.12 is apparently surprising: looking at the plant level data, it is not clearly observed a significant change in the energy mix in the established firms or a reduction in the emission factors of the different energy sources. Therefore, the factors that lead to this result not depend on direct incentives of the EU ETS on investment choice in greener technology or innovation. Therefore, the switch observed in the EI paths is mostly driven by the new entrants: the cut of free allowances fostered the entry of low-carbon technology plants in the sectors that face a lower or no share of free allowances in the future. On the contrary, secotrs that kept a high share of free allowances experienced the entrance of dirty producers compared to the established ones.

The findings pertaining to the entire sample combined with the descriptive analysis of the previous section demonstrate that the alteration in emission intensity is primarily influenced by the new entrants within both groups. Figure 2.12 visually demonstrates a significant transition occurring at the onset of Phase III between the average emission intensity in the control group and in the treated group. the treated group consistently reduces its average emission intensity, while the control group experiences an increase in its value.

TABLE 2.1: Regressions results

VARIABLES	(Baseline) EmissInt	(Model 1) EmissInt	(Model 2) EmissInt
did	-0.0541***	-0.0864***	-0.0682**
	(0.0168)	(0.0321)	(0.0326)
T	0.201***		
	(0.0427)		
POST	0.0645***		
	(0.0143)		
new#T			-0.0857***
			(0.0280)
Constant	0.225***		
	(0.0330)		
Time FE		✓	✓
Sector FE		✓	✓
Observations	4,795	4,795	4,795
R-squared	0.023	0.294	0.295
Number of AUT	658	658	658

Standard errors in parentheses
*** $p<0.01$, ** $p<0.05$, * $p<0.1$

The Table 2.1 presented provides the estimates for the difference-in-differences regressions, with the dependent variable being the emission intensity (EmissInt). In the baseline, the coefficient for the treatment effect (did) is estimated to be -0.0541, indicating a significant difference between the control and treatment groups on emission intensity. These results suggest that the change in free allowances policy had a significant effect on emission intensity.[15]

Overall, these findings provide support for the hypothesis that the change in free allowances policy had a significant impact on emission intensity, particularly among the new entrants. Indeed, the inclusion of new entrants in the dataset can have a substantial impact on the results of the Difference-in-Differences regression analysis. New entrants, by their very nature, introduce a dynamic element to the study, potentially altering the treatment and control groups' characteristics over time. What it is observed from the sectoral analysis leads to the idea of the "reverse Porter hypothesis": a lenient environmental regulation not only does not foster innovation and competition benefits for the firms, but fosters the entry of dirty producers in the sector.

When considering the impact of policy changes, such as the adjustment in free allowances provision, the presence of new entrants adds complexity to the evaluation. These entrants might be subject to different conditions, incentives, and trajectories compared to the established firms. In the context of the EU ETS, these new entrants could have joined the system

[15]The findings are robust even when excluding the refinery sector and/or including thermoelectric plants.

with more recent knowledge about policy frameworks, technological advancements, and industry best practices. This can lead to differences in their emission intensity behaviors, production strategies, and overall performance. The observation of how new entrants influence the findings of the study is pivotal for a comprehensive analysis of the impact of policy changes. Figure 2.14 provides a visual representation of this influence. Within the treatment group, the newly entering plants exhibit lower emission intensity compared to their established counterparts. Several factors might have played a role.

Firstly, it's conceivable that these new entrants possess a heightened awareness of the policy landscape and the impending free allowances cut. Armed with this knowledge, they could have strategically aligned their production processes with the anticipated policy changes, resulting in a more environmentally efficient approach. Moreover, it is possible that the treated sectors were poised for technological enhancements that could be swiftly incorporated by new entrants, the observed lower emission intensity might be a manifestation of these readily implementable improvements.

On the contrary, within the control group, where no substantial reduction in free allowances was experienced, the dynamic between new entrants and established plants appears to diverge. Here, the emission intensity of new entrants tends to be skewed towards the higher end of the spectrum within their group. This discrepancy might be due to a range of factors such as a lack of immediate incentive to adopt cleaner technologies, different strategic goals, or even structural limitations that hinder rapid improvements in environmental performance.

These observations underscore the importance of accounting for the heterogeneous characteristics of new entrants when evaluating the effects of policy changes. Such considerations should be a fundamental component of the interpretation and implications of this study. This analysis contributes to a clearer understanding of how policy interventions interact with the evolving industrial landscape, shedding light on both intended and unintended consequences.

2.5.2 Balanced sample: EU ETS plants since 2005

An interesting research question is whether an increasing (decreasing) emission intensity is primarily driven by the efficiency (inefficiency) of new entrants, or if it stems from innovation within the plants that have been participants in the EU ETS since its inception. To address this question, it becomes imperative to examine a subset of the sample comprising plants that have remained within the EU ETS framework since its inception and are still operational today. In this work, this sample belongs to the "established" group. By doing so, it becomes feasible to disentangle the effects of potential enhancements in carbon intensity among ETS "established" plants, and to discern the influence of new entrants on this metric.

This subset comprises a total of 129 plants, with 87 falling within the treated group and 41 within the control group. It is important to note that observations for the thermoelectric and textile sectors are absent, and the food industry includes only one plant with complete data for the entire period, leading to its exclusion from the analysis. This refined sample ensures balance, encompassing observations spanning from 2005 to 2019, with the exception of 2006 and 2007 due to data gaps and under-reporting in the Italian Registry.

The balanced sample is a subset that excludes sectors lacking continuous plant data from 2005 to 2019 (such as thermoelectric and textile sectors). Additionally, the food industry sector, having only a single observation, was omitted for privacy considerations. It is evident from the figure that a significant proportion of the sample pertains to the paper industry, contributing 71 observations per year.

The behaviour exhibited by each sector concerning the components of interest holds paramount significance in comprehending the underlying mechanisms inherent to the analysis of the entire sample. Figure 2.13 provides an account of the sector-specific trends in emission intensity, quantity produced, and emissions within the established plant sample.

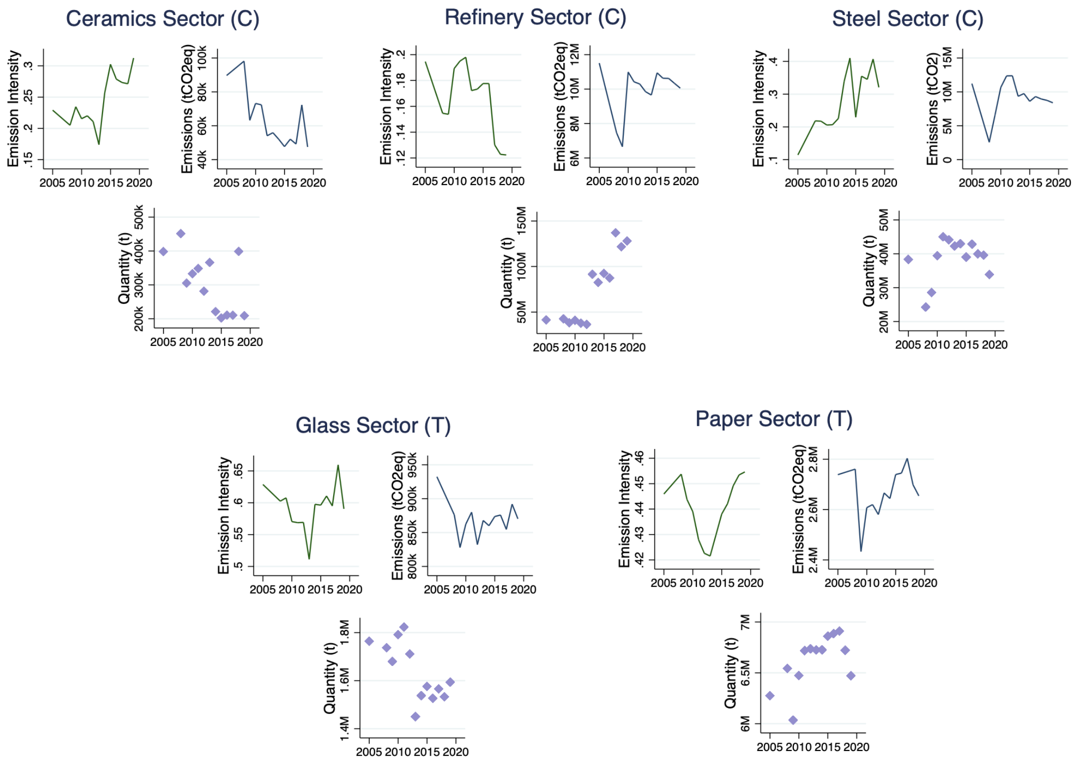

FIGURE 2.13: Description of balanced sample for each sector.

The sample is divided into control and treated plants. Control plants are those in sectors that did not experience a reduction in free allowances provision in the Phase III due to the amendment. Within the control group, the ceramics sector saw a decline in both emissions and quantity produced, subsequently leading to an increase in the average emission intensity over time. A similar trajectory is observed for the emission intensity of steel plants, which shows a slight upward trend. Notably, the refinery sector stands out as the top performer in the control group, showcasing a decreasing emission intensity trend, largely attributed to the concurrent increase in quantity produced. In the treated group, the emission intensity demonstrates a negative peak between the years 2012 and 2015, followed by a relatively stable pattern. However, the drivers behind these trends differ between the two sectors. For glass plants, emissions initially decrease from 2005 to 2009 before stabilizing, alongside a concurrent decrease in quantity produced. Conversely, the paper sector exhibits a slight upward emission trend, aside from an initial drop, coupled with an increase in quantity up until 2017.

The sector-specific descriptive analysis performed so far, while informative, does not provide a clear indication of whether the reduction in free allowances within the treated sectors had a discernible impact on production, emissions, and emission intensity. The underlying question remains: did the treated plants manage to enhance their efficiency compared to other sectors that did not experience a change in free allowances provision? Maintaining a descriptive stance, the treated group of the established plants exhibits a consistent and stable average emission intensity over time, while in the control group there is an observable upward trend.

2.5.3 New entrants

Referring to the "new entrants", these are the plants that became part of the EU ETS in 2013. The subsequent analysis will involve a comparison between the average emission intensity of these new entrants and the established plants within both the treated and control groups. This analysis reveals one of the most compelling outcomes of this study: as depicted in Figure 2.14, the treated group of the new entrants exhibits lower emission intensity in comparison to the pre-existent Phase III plants within the same group. Conversely, the control group of the new entrants displays an emission intensity towards the higher end of the spectrum within their group. These findings align with the theoretical structure considered in the working paper by Ambec, Esposito, and Pacelli, 2023 (Chapter 1), suggesting that free allowances do not directly alter the incentives to abate emissions, but they facilitate the entry of environmentally less sustainable producers.

Who are the new entrants? The sectoral composition of the plants that opted into the EU ETS during Phase III follows this structure: approximately 38% of these new entrants belong to the agrifood sector, while textile accounts for 2%, paper for 7%, thermoelectric plants for 3%, steel for 21%, ceramics for 27%, and refinery and glass for 2%.

Compared to the sectoral composition of the balanced sample, the composition of new entrants appears to be well-distributed between the treated and control groups. Notably, ceramics and agrifood plants emerge as the primary categories of new entrants, followed by steel and paper production facilities.

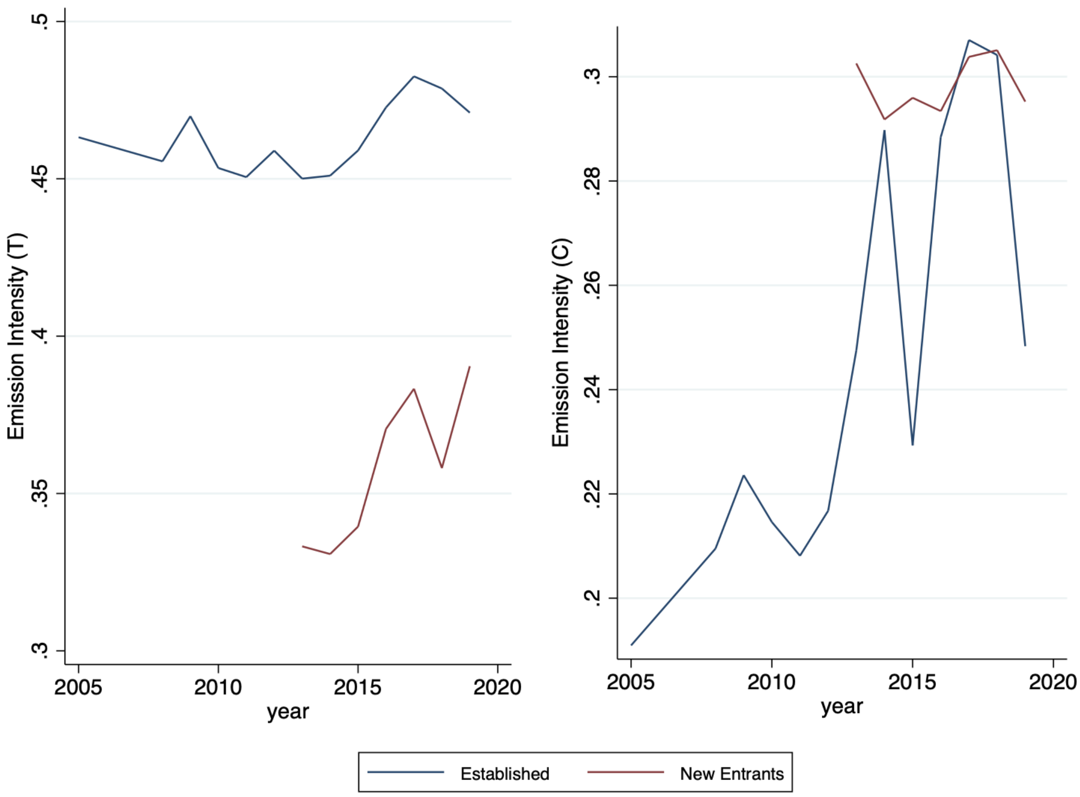

FIGURE 2.14: Established plants vs new entrants: behaviour of treated and control plants.

2.6 Conclusions

This study has empirically explored the Porter hypothesis in the EU ETS framework. Analysing the effects of the reduction in the provision of free allowances on emission intensity, the paper reveals that the change in free allowances provision had a significant impact on emission intensity, which is seen as a proxy of innovation towards low-carbon intensive production. Sectors that experienced a relevant drop in the free allowances provision slightly decrease their emission intensity, mainly due to the entrance of low-carbon emitting plants. This happens mainly in the agrifood sector, where the Porter hypothesis seems to be verified. In this sector, plants were able to decrease their emission intensity and new entrants are also more emission-efficient. On the contrary, sectors that kept a high share of free allowances do not abate their emissions and the new entrants are more emission-intensive. This is in line with what I defined the "reverse Porter hypothesis", meaning that a lenient environmental policy not only does not provide incentives to firms to innovate, but fosters the entry of dirtier producers.

Overall, this study highlights the importance of considering the heterogeneous characteristics of new entrants when assessing the effects of policy changes. It provides valuable insights into how policy interventions interact with the industrial landscape and contribute to both intended and unintended consequences. These findings have implications for policymakers and industry stakeholders seeking to achieve environmental goals within emissions trading systems.

Future researches on the EU ETS want to go further on the role of free allowances and their current phase out option pursued by the institutional authorities. The use of free allowances to avoid carbon leakage has raised concerns regarding the effectiveness of the EU ETS. The objective of the European Commission and Parliament is to avoid the issuance of free allowances and rely in the next future on other carbon leakage mitigation policy tools, such as the Carbon Border Adjustment Mechanism (CBAM). How to structure the CBAM, its interaction with free allowances and export rebates is the object of a large literature (Fischer and Fox, 2012, Böhringer et al., 2022, Ambec, Esposito, and Pacelli, 2023).

The current phase is the Phase IV (2021-2030) which also faced novelty and amendments: among those, an increased reduction of the cap (from 1.74% to 2.2% per year) and the gradual phase out of free allocations. The current price of the EU ETS allowance is 80 euros per ton. The Phase IV will be therefore crucial in understanding the phase out option effects and the CBAM mechanisms. Additionally, the Phase IV is going to be an interesting object for research due to the price level: from 2020, the price has an increasing trend, reaching about 105 euros per ton in March 2023.

Appendix

Additional Figures

Figure 2.15 is the same of Figure 2.12 but with two y-axes.

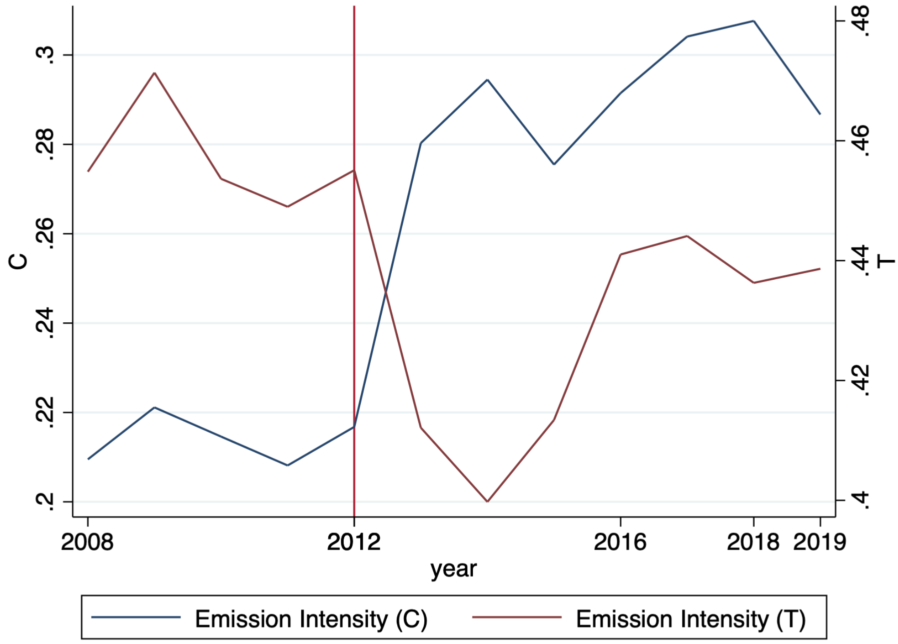

FIGURE 2.15: Emission Intensity, treated and control group (excluding thermoelectric).

Figure 2.16 illustrates the distribution within the balanced sample (left and the new entrants, opted in in 2013 (right).

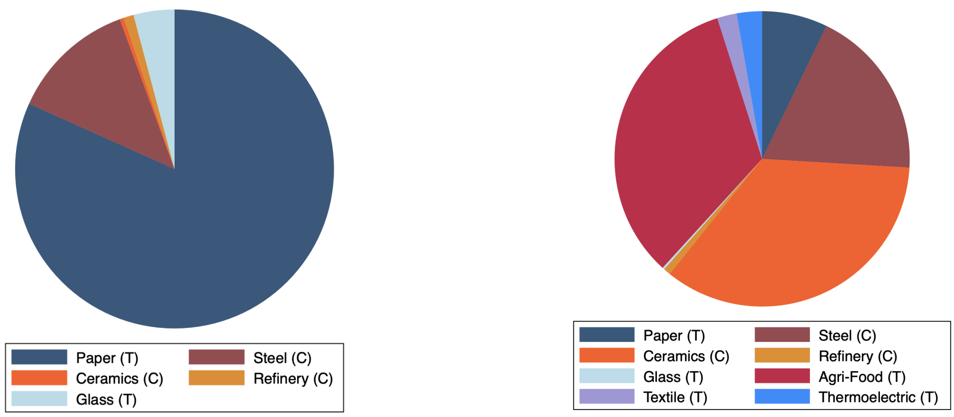

FIGURE 2.16: Sectoral composition of the balanced sample and of the new entrants.

FIGURE 2.17: Sectoral composition of new entrants.

Thermoelectric case

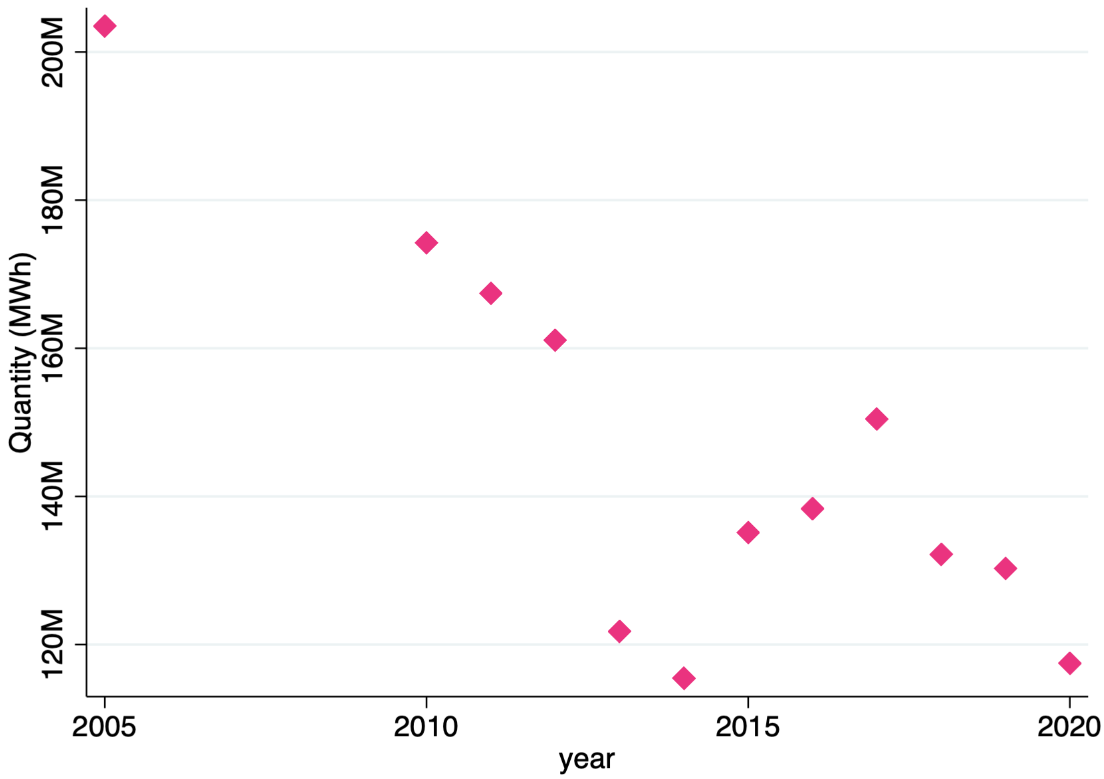

FIGURE 2.18: Quantities (MWh) for EU ETS Italian thermoelectric plants.

The quantity produced is decreasing over time, as showed in Figure 2.18. The main jump is in 2013, when 40 out of 140 electric plants opted out.

Figure 2.20 considers also the thermoelectric plants when computes the emission intensity average per year. Compared to Figure 2.10, it has slightly higher values and a similar decreasing path.

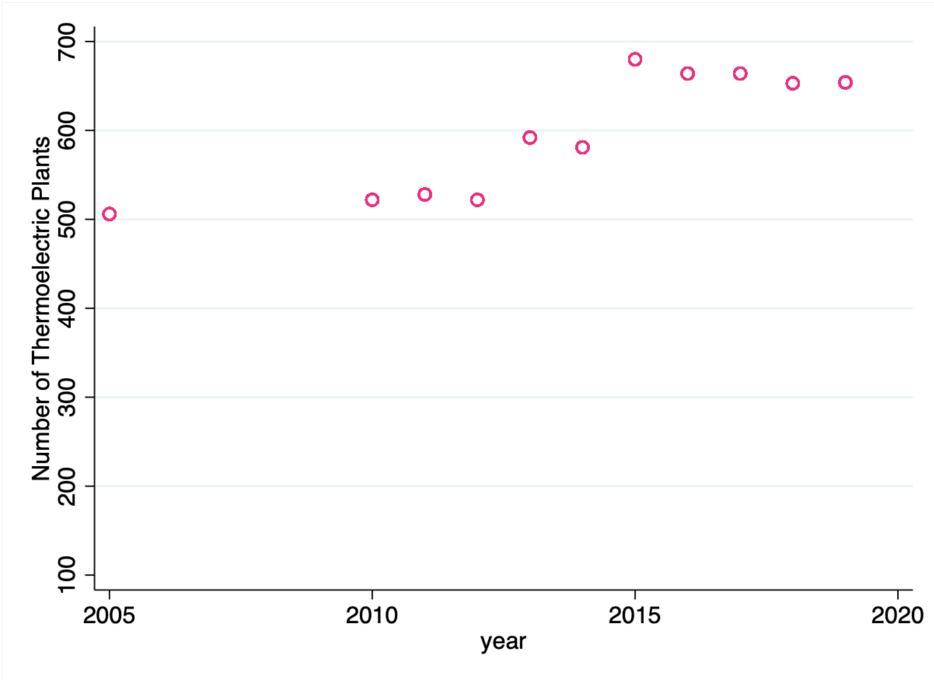

FIGURE 2.19: Number of plants in the thermoelectric sector.

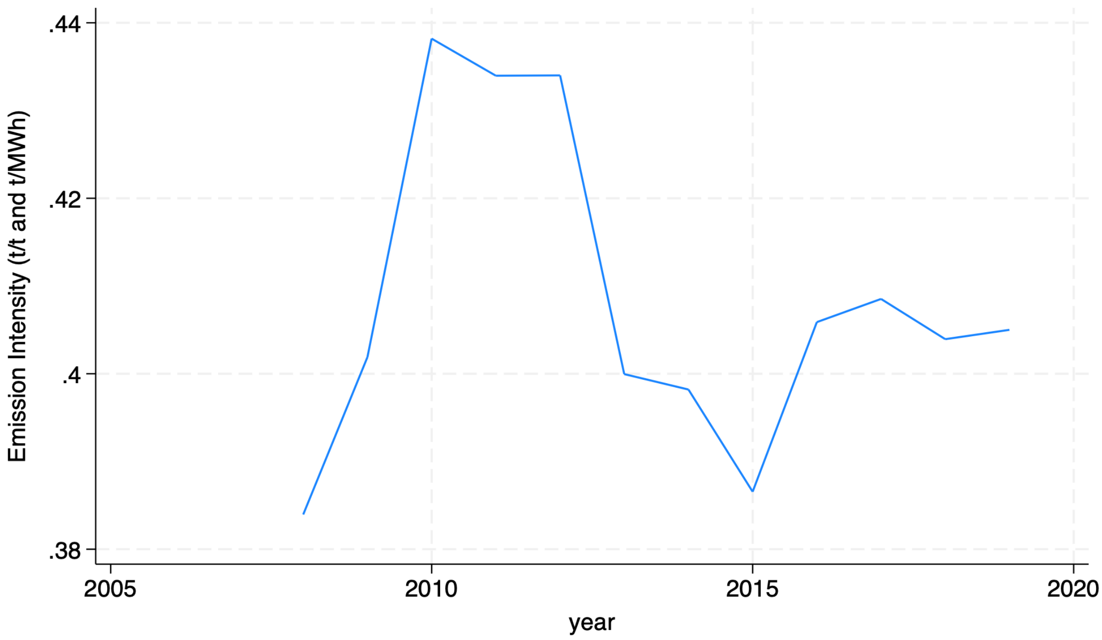

FIGURE 2.20: Emission Intensity considering also thermoelectric plants.

Additional Tables

Sector	**NACE (Rev.2)**
agrifood	10;11
Ceramics	23 (excl. 23.1;23.5)
Glass	23.1
Paper and Wood	16;17;18
Refinery	19
Steel	24-25
Textile	13
Thermoelectric	35

TABLE 2.2: Sectors and NACE codes.

Chapter 3

A European Climate Bond

Irene Monasterolo[1], Antonia Pacelli, Marco Pagano[2] and Carmine Russo[3]

The European Union faces a large climate investment gap. To fill it, we propose the joint issuance of EU climate bonds. These bonds would be funded by the sale of emission allowances, traded on the EU Emissions Trading System and extended to cover all sectors. Access to the resulting funds would be conditional on countries' performance on the implementation of climate investments. EU climate bonds would meet global demand for a safe and liquid asset, while increasing the speed and efficiency of EU climate investing, its resilience to sovereign crises, and the greening of investors' portfolios and monetary policy.

JEL classification:D62, E61, H23, H27, P18, Q51, Q52, Q53, Q54, Q58.
Keywords: climate finance, green investment, EU safe asset, emission allowances, ETS.

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3.1 Introduction

There is widespread consensus that the trend increase in greenhouse gas (GHG) emissions stemming from human activities is responsible for global warming (EC, 2022a; IPCC, 2022a).[4] This has triggered awareness of the urgency to limit GHG emissions, so as to avoid the potentially catastrophic effects of global warming. To this aim, in 2015, the United Nations (UN) approved the Paris Agreement, which set the target to limit global temperature increase to "well below 2°C above pre-industrial levels" (UNFCCC, 2015). However, recent analyses show that countries are not on track to deliver on this target, as most of them have not introduced yet stringent limits on GHG emissions (Conference of the Parties, 2021). Indeed, with the exception of the COVID-19 period, CO_2 emissions from fossil fuel combustion and related activities have increased so rapidly as to put the world on course for a temperature rise of 2.9°C by the end of this century (Burton and Muttitt, 2023). 2023 has been the hottest year on record, with a 1.46°C rise in the average world temperature relative to the pre-industrial level (Copernicus Climate Change Services, 2023), and the European Union (EU) already reaching 2°C on average in 2021 (Euractiv, 2022).

Missing the Paris Agreement temperature target is expected to entail large economic and social costs, due to increasingly frequent and severe disasters (climate physical risks), and to the structural adjustments of the economy required for fast decarbonization (climate transition risk).[5] Climate-related losses have already risen from \$895 billion (bn) in the 1978-1997 period to \$2,250bn in 1998-2017 – a 150% increase (UNDRR, 2018). Even under conservative estimates of future disasters and limiting the analysis to the euro-area countries, delaying the low-carbon transition after 2030 would lead to over 12% real GDP loss by 2050 compared to an orderly transition characterised by an early and smooth carbon tax introduction (Gourdel et al., 2022). From 2014 to 2022, Europe has already experienced a substantial increase in a range of adverse effects of climate change, including heat waves, changing precipitation patterns, sea-level rise, and increased frequency and intensity of extreme weather events (IPCC, 2022a). The economic costs are likely to be even larger in developing countries, which can invest fewer resources in

[4]While in the pre-industrial age (i.e., before 1900) the CO_2 concentration was on average 278 ppm (parts per million), it has steadily increased thereafter, especially since 1960 (320 ppm), reaching 424 ppm in May 2023. Over the same interval, the global temperature has feature a similar long-run accelerating increase: it rose by an average 0.08° C per decade from 1880 to 1981, and by an average 0.18° C per decade from 1981 to 2022 (NOAA, 2022).

[5]See also NGFS, 2019; ECB, 2020 and BIS, 2021.

climate action and are generally more exposed than European countries to natural disasters, such as floods, droughts and sea-level rise.[6]

However, mitigating climate change and adapting to it requires large investments and coherent policies (IPCC, 2022b). Recent estimates by the Climate Policy Initiative (Buchner et al., 2021) place the funding needs in the range of 4.5 to 5 trillion (tn) dollars per year at the global level, mostly for investments in transport, energy systems and efficiency.[7] In the EU, climate investment needs range between €550bn and €912bn per year.[8] Nevertheless, such estimates mostly cover climate mitigation investments, which are aimed at reducing GHG emissions. Instead, the financing of climate adaptation investments, which are meant to build resilience to climate change, is still largely neglected and imprecisely estimated.

Current plans by the European Commission (EC) go partly towards filling these investment needs, with ambitious programs such as the Green Deal and the Next Generation EU (EC, 2019; EC, 2022d).[9] Nevertheless, the resources currently budgeted for climate investment programs by the EU and its member states (MS) cover less than half of the investment needs in this area (see Section 3.2 below). Moreover, EU climate investment programs are threatened by policy uncertainty (Battiston et al., 2021), including changing green standards, the potential weakening of green regulation, and the reallocation of climate funding to other priorities.[10] They are also hindered by the lack of a common and coherent fiscal framework and by differences in fiscal and implementation capacity across MS. In addition, the new EU fiscal rules will tighten fiscal space of MS, implying likely reductions in their ability to invest in climate mitigation and adaptation.[11]

To address the EU climate finance investment gap, in this paper we propose a joint climate debt financing scheme aimed at addressing EU climate mitigation and adaptation needs. Our proposal consists of three complementary policy reforms: (i) the introduction of a uniform

[6] For country-level data: World Bank (2023a).

[7] These investment needs greatly exceed those currently undertaken. Global mitigation investments amounted to approximately $1.3tn in 2021-22 (IPCC, 2022b). Adaptation investments have reached $63bn worldwide (Buchner et al., 2023).

[8] For details see: EC, 2021a; EC, 2022b; EC, 2022c

[9] The EC has also undertaken relevant actions regarding climate disclosure (e.g. EC, 2020) and climate risk assessment, including climate scenario analysis and climate stress-test (ECB, 2022; EIOPA, 2022).

[10] See e.g. Gavin et al., 2023; Taylor, 2021.

[11] The agreement includes benchmark thresholds for all countries for annual average reduction of one percentage point in the debt ratio for countries with debt above 90% (e.g. Italy, France and Spain) and 0.5% for those between 60% and 90%, and a structural deficit margin of 1.5% of GDP, see EC, 1997.

EU carbon pricing scheme, (ii) the joint issuance of EU climate bonds, to be serviced by the revenues of the common carbon pricing scheme, and (iii) the implementation of a EU climate policy plan funded by issuance of these bonds.

The EU carbon pricing scheme is to be based on the sale of carbon emission allowances that polluting households and firms in all industries are required to purchase to be compliant with EU regulation. This scheme is simply an extension of the existing EU Emission Trading System (ETS) to all sectors, in line with current plans by the EC.[12] This scheme would substitute for a EU-wide carbon tax, as it would face all EU firms and households with a common carbon price, equal to the ETS permit price. By fine-tuning its sales of ETS permits, the EC would be able to calibrate this EU-wide carbon price at a level consistent with a science-based target. Moreover, the ETS permit sales would enable the Commission to directly appropriate the resulting revenue: compared to a system of national carbon taxes, this scheme would not be at the mercy of national choices regarding carbon tax rates and of MS decisions to transfer the resulting tax revenue to the EC. As such, ETS permit sales effectively represent the first form of EU common fiscal policy. Moreover, the EU-level commitment to a future path of carbon prices will contribute to reduce transition risk, anchoring investors' and firms' expectations (Fuest and Meier, 2023). Importantly, it would prevent the weakening of mitigation targets driven by political instability at national level and by the capture of MS governments by pressure groups.

We compute the present discounted value (PDV) of the revenues that such a EU-level carbon pricing scheme will generate under alternative scenarios for the future path of carbon prices and revenues, based on the climate mitigation scenarios provided by the Network for Greening the Financial System (Menon, Holthausen, and Breeden, 2022). Even in the scenario associated with the lowest possible value of the PDV (equal to €2.2tn), the resulting fiscal capacity would be of the same order of magnitude as the funding already budgeted for EU climate policies (EC, 2021c). In contrast, in the scenario where climate policies will remain fragmented across the world's regions, the fiscal capacity resulting from the proposed scheme would be much larger (€11.5tn), and could fill the EU climate financing gap for many years.

[12] Currently, this requirement does not yet encompass all sectors: it only applies to EU firms belonging to the sectors listed in Annex 1 of the Directive 87/2003/EC (EC, 2003). However, the EU Commission already envisages the extension of ETS emission allowances (so-called ETS 2) to fuel combustion in buildings, road transport and small industries, which are not covered by the existing ETS (EC, 2023a).

The sale of ETS allowances will enable the EC to service EU climate bonds. Issuance of these bonds will provide immediate access to these future carbon revenues, avoiding the postponement of urgent climate investments in all EU MS. We expect the servicing of these bonds to be cheaper than that of bonds issued via the NextGenEU program, due to their regular issuance, greater scale, and exclusive objective of financing "green" investments. The EU climate bond will also contribute to meet the current demand for a EU safe asset, which would form the backbone of an integrated European capital market, and thus contribute to the accomplishment of the long-awaited Capital Market Union (CMU) in Europe. At the same time, the issuance of such bonds would help global investors to green their portfolios. Finally, it may prove a valuable addition to the portfolio of assets held and traded by the European Central Bank (ECB) in the conduct of its monetary policy.

A novel feature of our proposal is the coordination between EU-level fiscal and climate policies. Beside creating a federal fiscal capacity to jointly fund climate investments in Europe, it would bring discipline to the design and implementation of these investments, by tying countries to a system of check and balances on the use of the funds raised via the EU climate bonds. Indeed, in our proposal, access to the funds raised by issuing EU climate bonds will be conditional on countries' performance on the implementation of the planned climate investments. Projects' delivery will be evaluated against the achievement of a set of Key Performance Indicators (KPIs), to be supervised by the European Investment Bank (EIB).

The paper is organized as follows. Section 3.2 quantifies the EU climate finance investment gap, considering both mitigation and adaptation policies. Section 3.3 discusses why it is more efficient to design and fund climate policies at the EU level rather than at the national level. Section 3.4 presents our policy proposal, and estimates the EU climate bond's issuance capacity based on the climate scenarios generated by the Network for Greening the Financial System (NGFS) for the EU. Section 3.5 discusses the macroeconomic and financial benefits from the issuance of EU climate bonds. Section 3.6 presents concluding remarks.

3.2 The EU climate investment gap

Climate investments have two distinct but interconnected objectives, i.e. mitigation and adaptation. Mitigation investments aim at preventing or decreasing the release of pollutants that contribute to climate change, for instance replacing fossil-fueled energy production with nuclear, solar, wind and geothermal energy plants, and connecting these to power-hungry, densely populated areas by suitably extending the power grid. In contrast, adaptation investments aim to increase the resilience of the economy to the effects of climate change, e.g. by protecting coastal areas against sea-level rise and areas exposed to the risk of floods, wildfires and landslides. Mitigation and adaptation investments are complementary: reducing emissions via earlier and more effective mitigation results in lower global temperature increase, and therefore in lower incidence and costs of climate-related natural disasters. Hence, both types of investments are needed to increase societal resilience to climate change.

Climate investments are usually funded by general taxation or by carbon taxes, as well as the revenues from the sale of emission allowances such as the ETS in the EU. They are also funded via the sale of green bonds, whose proceeds are earmarked to finance investments in renewable energy, transportation and construction industries.[13]

The EU mitigation investment needs over the period 2020-30 include €58.4 bn per year to be invested in the electric grid and €336 bn per year for energy system investments, excluding transport (EC, 2021a; EC, 2022c), while estimates of adaptation needs vary widely, ranging from €158 to 518 bn/year (EC, 2022b). Based on available estimates, the sum of EU mitigation and adaptation needs ranges between €550bn/year and €912bn/year (EC, 2021a; EC, 2022b; EC, 2022c). The official EU estimate is in the middle of this range: the EU-27 MS and the EC must invest over €700 bn per year to achieve the Green Deal target of Net Zero emissions by 2050 (EC, 2023d).

The resources currently budgeted by the EU and its MS for their climate policies in 2021-27 fall short of these investment needs. The EC long-term budget of €2tn at current prices (30% of EU budget) implies spending about €330 bn/year for mitigation, adaptation and cost of natural disasters (EC, 2021c). In addition, within the Recovery and Resilience Facility Programs, EU

[13] See EC, 2023b.

MS were required to allocate at least 37% of spending to climate investments (EC, 2021b). This leaves a sizeable gap between EU climate investments needs and budgeted expenses. Based on the EU official estimates of its investment needs (€700 bn/year), this "climate investment gap" amounts to

$$\text{investment gap} = \underbrace{\text{investment needs}}_{€700\text{bn/y}} - \underbrace{\text{budgeted expenses}}_{€330\text{bn/y}} = €370\text{bn/y}$$

Based on the €912bn/year upper bound of climate investment needs, the gap would rise to €582bn/year. In fact the gap may be even larger, considering that these estimates may still omit mitigation and adaptation needs that we are unaware of, given the uncertainty associated to climate impacts.

3.3 Designing and funding EU climate policies

In principle, the EU climate investment gap may be partly covered by national MS budgets. However, in 2019 EU MS only spent €90bn on climate investments (OECD, 2022), namely, less than one quarter of the investment gap. This highlights the challenges of financing climate investments at the national level.

Moreover, recent experiences of financing climate investments by individual MS are quite heterogeneous, with some countries being more active, and others being unable to allocate adequate resources to climate policies out of their national budgets and to implement climate investment projects, such as those envisaged by the NextGenEU program (EC, 2021c).

Hence, designing and enforcing climate policies at the supra-national level could not only contribute to increase overall spending on climate actions within the EU, but also reduce inefficiencies due to their cross-country heterogeneity, as described in Section 3.3.1. EU institutions are obvious candidates to be entrusted with the design and enforcement of joint climate policies. But such enforcement is naturally much more effective if the EU were also to raise and allocate the funding required for such policies, as discussed in Section 3.3.2.

3.3.1 Why climate policies should be designed at the EU level

Designing climate policies at the national level can generate inefficiencies for several reasons, namely, (i) spatial spillovers, (ii) regulatory externalities and (iii) regulatory capture.

First, carbon emission spillovers across national borders imply that individual MS may opt for too lenient environmental targets, simply because the resulting harm would be partly borne by neighboring countries.

Second, polluting firms can choose across different jurisdictions by relocating their activities across national borders ("emissions offshoring"), i.e., may engage in regulatory arbitrage. For instance, increases in domestic fossil fuel prices resulting from national carbon taxes, or more stringent emissions targets, may lead to the re-allocation of production to countries with less stringent mitigation rules–a phenomenon known as "carbon leakage" (Ambec, Esposito, and Pacelli, 2023; Benincasa, Kabas, and Ongena, 2022; Laeven and Popov, 2022). In turn, this may induce governments to set inefficiently low national climate standards: each government has little incentive to introduce ambitious climate policies and regulations, fearing that these would induce domestic producers to relocate to more lenient jurisdictions and/or provide an advantage to foreign producers located there (Felder and Rutherford, 1993; Hoel, 1991). As a result, regulatory arbitrage also tends to induce a "run to the bottom" in national environmental standards. [14]

Third, even if climate policy standards are designed at the supra-national level, a similar "run to the bottom" may arise in the enforcement of the common standards if it is left to national governments. Insofar as national authorities are captured by domestic pressure groups and lobbies, they will tend to water down the enforcement of climate policies and regulations within their respective jurisdictions. Here a fitting parallel can be drawn with prudential bank supervision in the euro area: the design of common rules in banking supervision for euro-area banks has been supplemented by common enforcement by the Single Supervisory Mechanism

[14] It is precisely to avoid carbon leakage and distortions in international trade that the EC is currently planning the introduction of the "carbon border adjustment mechanism" (CBAM). This mechanism consists of imposing tariffs on imports so as to create a level playing field between domestic and foreign producers in the carbon price that they face. The EU is planning to introduce the CBAM in 2026 within the ETS. Initially, the CBAM will apply to imports of select industries (aluminum, cement, fertilizers, iron and steel, electricity and hydrogen), by charging to importers of these goods a carbon tax equal to the average price of permits traded in the ETS (EC, 2023c).

(SSM), recognizing that national central banks might otherwise be too lenient in their supervisory role. Entrusting climate policy standards to a supra-national authority would also shield these standards from the vagaries of national politics in countries featuring high government instability, thus increasing their credibility over time.

Designing climate policies at the supra-national level can address these inefficiencies, by settings the standards and objectives of climate policies exclusively on the basis of their contribution to decarbonization, while leaving their implementation to MS, as done under the current EU climate strategy in connection with the NextGenEU, consistently with the subsidiarity principle. The EU would however monitor the implementation of climate policies and projects by MS, by setting KPIs and entrusting their enforcement to the EIB, so as to overcome possible inefficiencies and moral hazard issues. To ensure incentive compatibility of this scheme, the EU can threaten to withhold further funding of non-compliant MS, again in line with the NextGenEU program.

3.3.2 Why climate policies should be funded at the EU level

As already noted, limited fiscal space may constrain public financing of climate policies. This already currently applies to EU MS featuring high public debt and elevated cost of debt service, but the fiscal constraint on their climate policies is likely to become even more stringent after 2024, once the Stability and Growth Pact is reinstated. This will require large fiscal adjustments by several countries and currently does not yet contain any exemption for green investment (i.e. "green golden rules"), as highlighted by Zettelmeyer, 2023.

The resulting under-investment in climate policies in some EU MS is likely to have negative spillover effects for other EU MS. First, there may be physical spillovers via greater cross-border emissions, as already noted in Section 3.3.1. Second, under-investment in climate policies by high-debt countries would increase their exposure to natural disasters, weakening not only their own economic performance, but also that of other EU MS via demand and supply chains. For instance, more fragile countries would tend to import less from the rest of the EU, and would contribute fewer exports of intermediate goods to foreign production. These spillover effects could be amplified by financial market reactions: investors may respond to increased climate risk in the affected countries by repricing their sovereign debt and cutting back on lending. This may generate a sovereign debt crisis, with potentially destabilising effects for other EU countries and for the common currency.

Therefore, allocating resources across EU MS so as to enable also the more vulnerable MS to fund climate policies is not only in the interest of high-debt EU countries but in that of the EU as a whole. Moreover, relaxing this fiscal constraint is all the more important considering that most climate investments will be frontloaded, as earlier spending on climate policies is expected to achieve larger co-benefits and imply fewer GDP losses from natural disasters (Emambakhsh et al., 2023; Gourdel et al., 2022).

Thus, efficiency requires joint EU-level climate financing. This is in line with the growing consensus for the creation of an EU fiscal union to fund EU spending on common goods. In this regard, the former ECB President Mario Draghi recently stated that

> "Europe must now confront a host of supranational challenges that will require vast investments in a short time frame, including defence as well as the green transition and digitisation. As it stands, however, Europe neither has a federal strategy to finance them, nor can national policies take up the mantle [...] Without action, there is a serious risk that Europe underdelivers on its climate goals" (The Economist, October 2023).

Relatedly, over 100 EU economists signed the 2023 Manifesto for Europe (Almunia et al., 2023), which calls for a fundamental reform of the EU budget built on a permanent or, at least, recurrent central fiscal capacity to supply European Public Goods in the triple green, digital and social transition. Recently, the President of the ECB, Christine Lagarde, also highlighted that joint EU-level climate financing may have important distributional benefits, as she stressed the importance of "sharing the burden fairly" so as to mitigate the short-term costs and related backlashes of frontloading green investments (Lagarde, 2023).

3.4 The policy proposal

Our policy proposal consists of three complementary reforms: the introduction of a uniform EU-level carbon pricing scheme, the joint issuance of EU climate bonds to be serviced by the revenues of this scheme, and the design and implementation of a EU climate policy plan funded by the issuance of these bonds. Section 3.4.1 presents our proposed carbon pricing scheme, explaining how it differs from existing carbon pricing regimes in Europe. Section 3.4.2 explains how joint issuance of EU climate bonds would enable the EU to tap the additional fiscal capacity created by the EU carbon pricing scheme, and compares it to the issuance of NextGenEU bonds currently implemented by the EC. Finally, Section 3.4.3 presents estimates of the federal fiscal capacity created by our policy proposal, and thus of the potential issuance of EU climate bonds, under different scenarios for the future path of carbon prices and revenues. These are in turn based on the climate scenarios generated by the Network for Greening the Financial System (NGFS), using the REMIND-MagPie process-based Integrated Assessment Model.

3.4.1 A EU-wide carbon price

The EU already has the most advanced carbon pricing regime in the world (World Bank, 2023b). In particular, the EU ETS is the first and largest cap-and-trade system allowing firms to trade CO_2 equivalent emission permits. The market is formed by two segments: a primary market with auctions, where permits are sold by the EC to firms, and a secondary market where firms and financial intermediaries continuously trade outstanding allowances. This market generates a single carbon price for the whole of the EU at each point in time. Since firms in all the EU-27 MS are subject to the EU ETS directive (EC, 2003), they must all pay the carbon permit price determined by the ETS, which in 2023 amounted to €88 per ton. Currently, the revenues from the sale of ETS allowances (around €20-25bn per year) are rebated to the respective MS and to the European Environmental Agency (EEA) and are mainly used to support climated policies (European Enviroment Agency, 2023).

On top of this market-based system for carbon allowances, various EU countries also feature national carbon taxes, whose rates vary greatly across them and generally differ from the common ETS rate, as shown by Figure 3.1: in most countries, carbon taxes are levied at a considerably lower rate than the ETS carbon price, with Sweden being an exception. Moreover, the emission tax rate can differ across pollutants, with some countries featuring two different rates, shown in the figure as carbon tax (1) and carbon tax (2). For instance, Denmark features a carbon tax on fossil fuels at approximately €25 per ton, alongside with a carbon tax on fluorinated gases at approximately €20 per ton. Similarly, Finland has a dual-tiered carbon tax system, encompassing a tax of €78 per ton on transport fuels and around €53 per ton for other fossil fuels. Estonia, France, Latvia, Spain and Sweden, instead, feature a single carbon tax rate. Finally, Germany and Italy charge no carbon tax, although Germany has an additional ETS on heating and transport fuels since 2021.

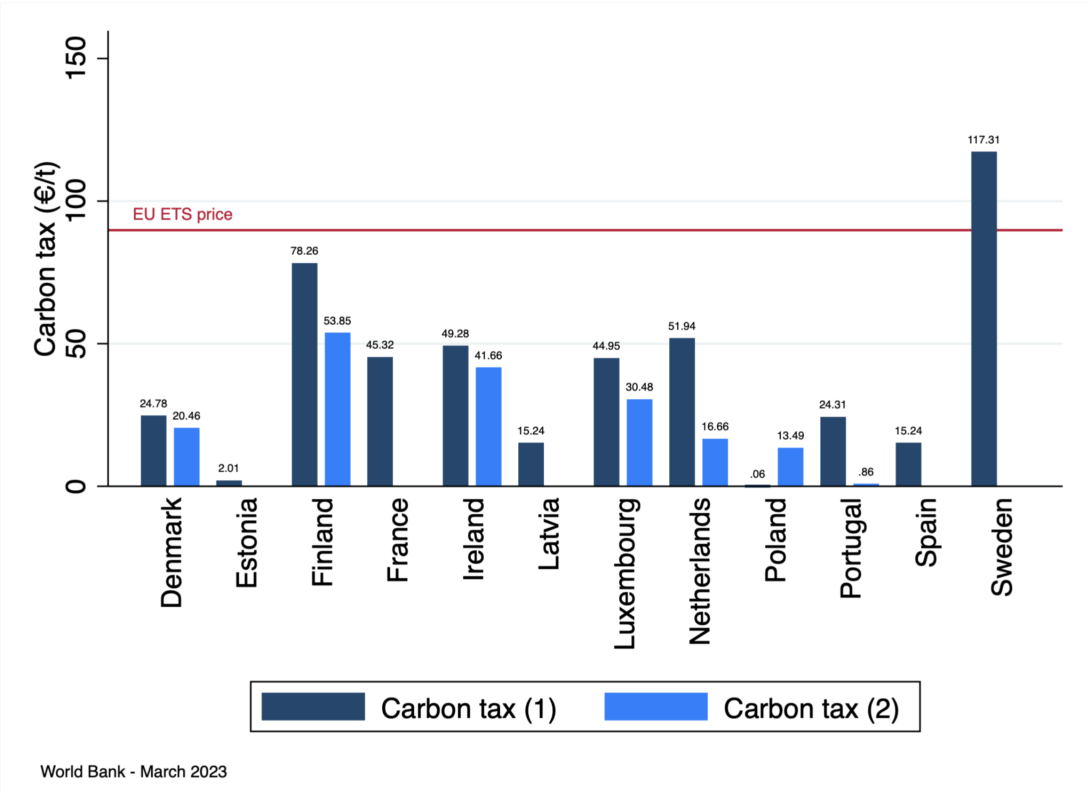


FIGURE 3.1: EU carbon tax rates

Both carbon taxes and ETS carbon allowances are policy instruments aimed at deterring GHG emissions. However, they differ in their characteristics and mechanisms. First, the allowances traded on the ETS set an upper bound on total carbon emissions and can be bought by firms depending on their needs, while carbon taxes place a price tag on emissions. Second, the ETS determines a single carbon price for the whole EU and applies to CO_2 emissions by all firms in a given sector (e.g., steel production), irrespective of the energy sources being used; in contrast, carbon taxes are set at potentially different levels by national member states and apply to specific sources of energy, such as fossil fuels, irrespective of the sectors in which they are used. However, firms in sectors required to buy ETS allowances face no carbon taxes. For instance, in France and Sweden there is no overlap between the two carbon pricing schemes, as firms in sectors required to buy ETS allowances (e.g., manufacturing firms) are exempted from the respective national carbon taxes (Government of Sweden, 2023). In France, where the ETS price exceeds the carbon tax rate, firms in industries required to comply with ETS allowances effectively pay a higher carbon price than firms in other sectors, while in Sweden the opposite occurs.

Our proposal aims at strengthening the current EU framework of carbon pricing by extending the requirement of ETS allowances to all sectors, so as to face all EU firms with a uniform and predictable carbon price. At each date, the EC can manage the supply of allowances available to firms so as to target a pre-announced, science-based path for ETS carbon prices, taking into account and smoothing temporary fluctuations in the demand for emission allowances. In principle, the EC can manage the supply of allowances both by changing the amounts sold in the primary market via auctions and by operating on the secondary market in the same way as central banks manage the money supply via open-market operations to target interest rates.[15]

A key aspect of our proposal is that the EC would retain the revenue resulting from the sale of ETS allowances within the EU budget (rather than rebating it to MS as currently done), effectively reallocating fiscal revenue from the state to the EC level and creating a source of federal tax revenue at the supranational level. However, this fiscal capacity would be deployed to fund climate policies designed and agreed at the EU level in the various MS according to climate risk priorities. Note that MS that generate more carbon emissions would contribute more to EU climate policies, as their firms and households would purchase a greater amount of ETS allowances; however, these MS would also benefit proportionately from spending on mitigation policies aimed at reducing future carbon emissions. This should ensure a rough long-term proportionality between the fiscal revenue contributed by each MS to this scheme and the funding it receives for its mitigation policies. However, some deviations from such

[15] Currently, the Market Stability Reserve (MSR), implemented since 2019, represents a long-term solution to address issues related to the supply of EU ETS allowances and their price (European Parliament, 2015). The surplus of allowances allocated during the initial two phases of the EU ETS and the consequent drop of the permit price required the creation of measures to regulate allowance supply. The MSR, designed to rectify over-allocation and to improve the system's resilience to major shocks, operates within the auction market to achieve its aims. However, it could be reasonable to extend the MSR's operations to the secondary market, so as to fulfill the role outlined in this proposal.

proportionality between contributions and spending across MS may be required to face adaptation investment needs (e.g., protection against sea-level rise or floods and hydro-geological erosion), which are likely to be disproportionately concentrated in some MS.

The proposed amendment to the current ETS would have three important implications. First, the new design of the EU ETS would be efficient, as it would face all emitters with a uniform carbon price, irrespective of their sector and national jurisdiction. Second, it would reduce transition risk in the EU, as firms and households would be able to base their investment decisions on a pre-announced target path for carbon prices. Third, since it would enable the EC to appropriate all the revenue stemming from carbon pricing in the EU via sales of ETS allowances, it will make national carbon taxes redundant. These revenues will provide the federal fiscal capacity needed to fund the issuance of EU climate bonds, as explained in the next section.

3.4.2 Issuance of EU climate bonds

EU climate bonds are to be jointly issued by a EU-level institution on behalf of all MS, and to be serviced them with the revenues from sales of ETS carbon allowances, as described above. Such a bond will appeal to investors for two main reasons.

First, EU climate bonds will enable investors with a sustainability mandate (e.g. Environmental Social Governance institutional investors, Net Zero alliance signatories, etc) to “green” their portfolios, because our proposal restricts the use of the revenue raised via their issuance to the exclusive funding of climate policies. This is envisaged to occur via conditionality clauses mandating precise criteria for the quality of the projects and via monitoring of their implementation through KPIs by the EIB. This should enable EU climate bonds to command a “greenium”, i.e., a lower yield on account of them being exclusively and credibly earmarked to support climate policies.[16]

[16] Currently there is no universal agreement in the literature regarding the existence of the greenium in the sovereign bonds market. Grzegorczyk and Wolff, 2022 and Baker et al., 2022 document a systematically lower yield for green sovereign bonds compared to traditional bonds, indicating a positive greenium. In contrast, according to Bolton et al., 2022, the evidence is consistent with a negative greenium, i.e., with the yield of green bonds exceeding that of comparable conventional bonds. They argue that this result hinges on the lack of credible, legally binding commitments from sovereign issuers to earmark funds for green projects, leading to investor distrust. However, as mentioned above, our proposal restricts the funds raised through the issuance of EU climate bonds to the funding of climate policies and envisages a mechanism to monitor their implementation. Hence, they should be able to command a greenium.

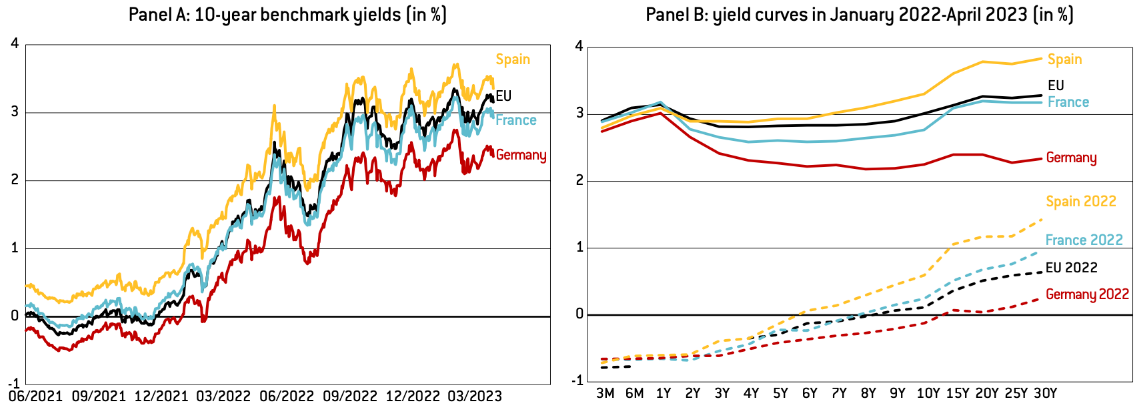
Second, the bond will be regarded by investors as a EU safe asset, on a par with national sovereign bonds, being directly backed by the revenue that the EC would obtain from sales of ETS allowances. As such, it would support a favorable treatment by prudential regulation of banks' and insurance companies' exposures, and would be used by the ECB as collateral in its monetary policy operations. If the issuance of these bonds is entrusted to the European Stability Mechanism (ESM), their repayment could also be guaranteed by unused ESM resources, so that the cost of servicing the bond would benefit from ESM's rating. Entrusting issuance of EU climate bonds to the ESM would also capitalize on the expertise and proved track record of an existing supranational institution in issuing bonds on behalf of the EU.

These characteristics would enable the issuance of EU climate bonds to overcome some of the limitations of the current issuance of EU bonds within the NextGenEU program in terms of borrowing costs and liquidity. This program allowed the EC to borrow up to €750bn by 2026, issuing bonds with maturities ranging from 3 to 30 years, based on a pre-agreed issuance volume, and placed mainly via bank-syndicated transactions. Moreover, no debt rollover was foreseen: bonds are to be repaid starting from 2028 up to 2058 (Grégory, Conor, and Lennard, 2023).[17]

Some weaknesses of the NextGenEU bonds emerge considering their funding cost: in 2023, their yields exceeded German ones by about 80 basis points (bp), and also French ones by about 20 bp, even though they were below their level in 2021 (Figure 3.2). This yield differential may reflect the comparatively low market liquidity of EU bonds: the bid-ask spread for 10-year NextGenEU bonds greatly exceeds that of the French and German bond with the same maturity, and recently also that of Spanish bonds, while their trading volume is way lower than that of these countries (Figure 3.3).

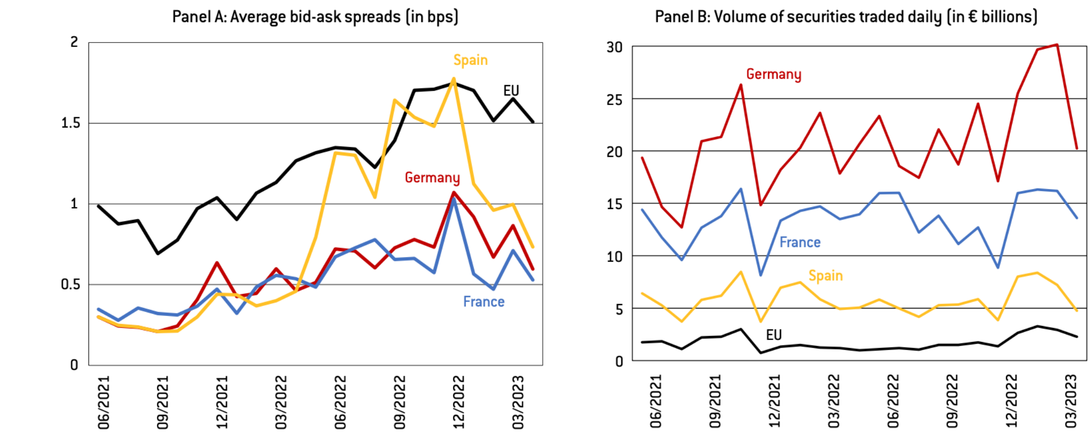
However, EU climate bonds are envisaged to differ from NextGenEU bonds in several important respects, making them far more appealing to investors and thus able to command lower yields (see Table 3.1):

[17] EU MS recently agreed to increase the EU's debt guarantees by adding 0.6% and might introduce new own EU resources in the future (European Parliament, 2021; Grégory, Conor, and Lennard, 2023).



Source: Bruegel based on Bloomberg. Notes: dashed lines represent data as of 3 January 2022 while unbroken lines represent data as of 11 April 2023. For January 2022, the EU yield curve was incomplete so the values for the 1- and 3-year maturity yields are extrapolated.

FIGURE 3.2: NextGenEU bond performance. Panel A shows the 10-year benchmark yield (in %), Panel B shows the yield curves between January 2022 - April 2023 (in %).



Source: Bruegel based on Bloomberg. Notes: Panel A: Monthly average of bid-ask spreads for 10-year bonds for selected issuers in basis points. Panel B: Monthly average of daily volume of security trades by issuer in € billions.

FIGURE 3.3: NextGenEU bond bid-ask spread and volumes. Panel A shows the average bid-ask spreads (bp), Panel B shows the volume of securities traded daily (bn€).

1. Being serviced by the predictable cash flow of sales of ETS carbon allowances for several decades, even long-maturity EU climate bonds could be rolled over several times and thus could be frequently issued according to a pre-announced regular calendar. This would guarantee a steady flow of freshly issued bonds, which are typically the most actively traded and liquid ones.[18] .

2. The scale of their total issuance would be from 3 to 15 times larger than that of the

[18] Krishnamurthy, 2002, Goldreich, Hanke, and Nath, 2005 and Goldstein and Hotchkiss, 2020 document not only that "on-the-run" bonds are more liquid than "off-the-run" ones with the same residual maturity, but that investors require a lower yield on them, reflecting a lower liquidity premium.

NextGenEU bond issuance (see Section 3.4.3 below). This should also contribute to making them more liquid than the NextGenEU bond, as larger asset issuance is well known to be associated with lower bid-ask spreads and higher turnover rates (Foucault, Pagano, and Röell, 2023).

3. While NextGenEU bonds are backed by MS via off-balance sheet items in their national budgets, EU climate bonds would be backed by ETS sales revenue flow directly appropriated by the EC via the sale of EU ETS. As such, it should be considered by investors as a EU-issued sovereign asset, rather than as a quasi-sovereign asset backed by national MS. This should enhance its perceived safety from investors' standpoint.

4. While NextGenEU bonds are issued to fund a variety of investment programs in MS, among which climate policies, EU climate bonds will be *solely* issued to fund climate investments. This will appeal to investors with a sustainability mandate, and should make it more likely that EU climate bonds will command a greenium.

5. Finally, while NextGenEU bonds are issued mainly via syndication procedures entrusted to a select group of large EU banks, the frequency and magnitude of EU climate bond issuance would warrant them being sold via regular auctions, which typically feature lower issuance costs than a syndication mechanism.

Next Generation EU bond	**EU climate bond**
fixed issuance → no rollover	regular issuance → debt rollover
low volume → low liquidity	high volume → high liquidity
backed by MS, off-balance sheet → quasi-sovereign asset → not fully safe asset	backed by ETS sales, in-balance sheet → sovereign asset → safe asset
funding various programs → no "greenium"	only funding climate policy → "greenium"
mainly placed via syndication → high issuance cost	placed via auction → low issuance cost

TABLE 3.1: Differences between NextGenEU bond and EU climate bonds

3.4.3 EU climate bond issuance capacity

The potential issuance of EU climate bonds will be determined by the fiscal capacity created by the EU-level carbon pricing scheme described in Section 3.4.1. To assess the magnitude of this fiscal capacity, we consider the carbon price and the Kyoto GHG emissions trajectories estimated for the EU27 by the NGFS under various climate mitigation scenarios, adapted from those reviewed by the IPCC (Menon, Holthausen, and Breeden, 2022).

The REMIND-MAgPIE process-based Integrated Assessment Model (IAM) provides estimates of the trajectories of carbon prices and of the corresponding production level in the EU consistent with a given temperature target, e.g, below 2°C (Kriegler et al., 2013). The model performs such estimation for a variety of scenarios, each of which represents a different type of transition.Here we use the estimated trajectories in four scenarios.[19]

The structure of the REMIND-MagPie model is presented in Figure 3.4. It includes a macroeconomic module connected to a land use module (MagPie), informed by a vegetation module and energy system module, which is in turn connected to a climate system module (MAGICC). The model translates climate scenarios into adjustments to production levels, considering their energy technology and impact on climate. The macroeconomic model establishes energy demand (considering variables such as population growth), while the energy model calculates energy supply and associated input costs based on a specified emission level and corresponding carbon price. The projected emissions pathways are used to estimate global temperature outcomes using the MAGICC model (Meinshausen, Raper, and Wigley, 2011).[20]

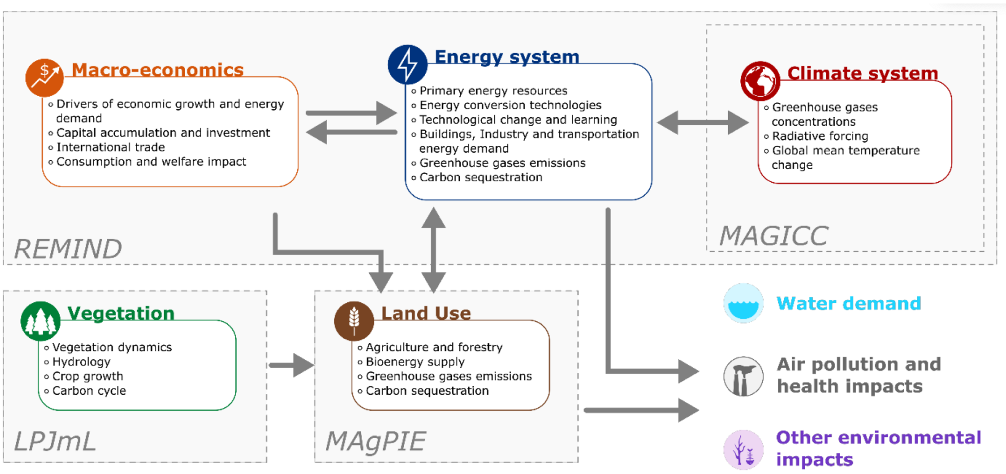


FIGURE 3.4: REMIND-MagPie framework

The four NGFS scenarios that we consider in our analysis run until 2100 and differ in terms of temperature target, timing and characteristics of climate action.[21] These distinctions translate

[19]Notice that the REMIND-MAgPIE model produces estimates for EU28 (Richters et al., 2022). EU27 Kyoto GHG emissions are obtained by subtracting the UK emissions from the EU28.

[20]The climate scenarios developed by process-based IAMs have been used for climate financial risk assessment since the climate stress test by Battiston et al., 2017. Investors in several jurisdictions (e.g. Euro-area banks and insurance companies) are required by their supervisory authorities to run climate stress tests using these scenarios (see e.g. ECB, 2022).

[21]For further details, see Ravi and Livio, 2023; Richters et al., 2022; Thomas et al., 2020.

in different levels of climate physical risk and transition risk, as illustrated in Figure 3.5:

1. The "delayed transition" scenario assumes a late and sudden introduction of climate policies, so that annual emissions do not decrease until 2030. The scenario features high transition risks due to delayed and hence costlier climate policies. These policies however limit physical risk by keeping temperatures below 2°C by the end of the century.

2. The "fragmented world" scenario assumes a delayed and divergent climate policy response among countries globally, leading to an increase in global temperatures around 2.3°C by the end of the century, and therefore to high physical and transition risks.

3. The "below 2°C" scenario considers an early introduction of climate policies that gradually increase in stringency implying a 67% chance of reaching the 2°C target. Thus, it is associated with both low transition and physical risk.

4. The "current policies" scenario assumes that climate policies are held at the currently implemented level, leading to low transition risk due to the absence of stringent climate policies, but high physical risk due to inadequate mitigation and adaptation policies.

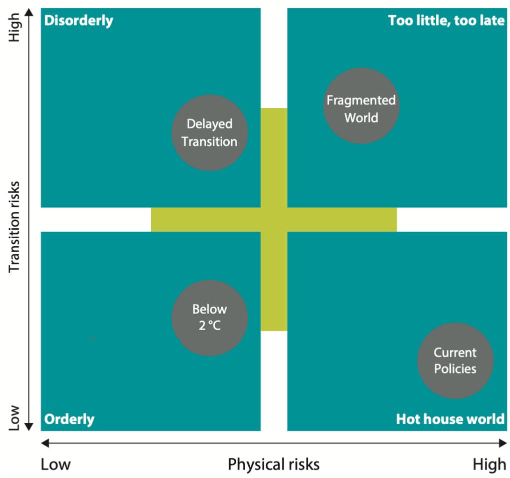


FIGURE 3.5: Selected NGFS Scenarios

For each of these scenarios, Figure 3.6 shows the estimates of the carbon price (in 2010 US dollars/ton), i.e., the shadow price of the cost-minimisation procedure necessary to reach the relevant target emission level (Gourdel et al., 2022), and the CO_2 equivalent Kyoto GHG (CO_2, CH_4, N_2O and F-Gases) emissions in Megatons (Mt).

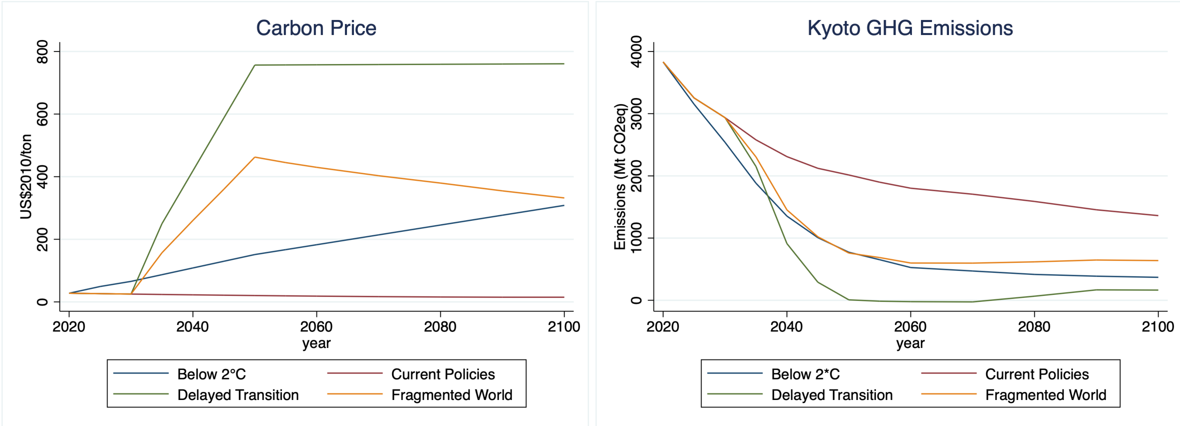


FIGURE 3.6: NGFS Scenarios Trajectories

The "delayed transition" scenario features the highest carbon price path and, consequently, the lowest level of GHG emissions by 2050. In contrast, in the "current policies" scenario the carbon price trajectory is flat, and as a result GHG emissions are the highest and most persistent. In the "below 2°C" and "fragmented world" scenarios the path of GHG emissions is comprised between these two extremes, but in the latter the carbon price and emission paths are more unstable than in former, as the carbon price stays too low in the first decade and must therefore rise sharply in the subsequent two decades.

Assuming that the EC manages the supply of ETS carbon allowances by targeting the carbon price estimated for each of the four scenarios, the revenues accruing to the EC will equal the product of the respective carbon price (in 2010 US dollars) and the corresponding Kyoto GHG emissions, upon converting them from megaton (Mt) to ton (t):

$$\text{estimated revenues} = \text{carbon price} \times \text{CO}_2\text{eq Kyoto GHG emissions.}$$

Figure 3.7 plots the resulting revenue trajectories for each of the four NGFS scenarios. In all four scenarios, revenues from the sales of ETS allowances are estimated to stay quite sustained and stable until the end of the century, as the change in quantities (emissions) is foreseen to be compensated by the change in carbon prices in the opposite direction. In most decades, revenues are projected to be highest in the "fragmented world" scenario and lowest in the "current policies" scenario. The path of carbon revenues is foreseen to be at an intermediate and stable level in the "below 2°C" scenario, while it is unstable with a "delayed transition".

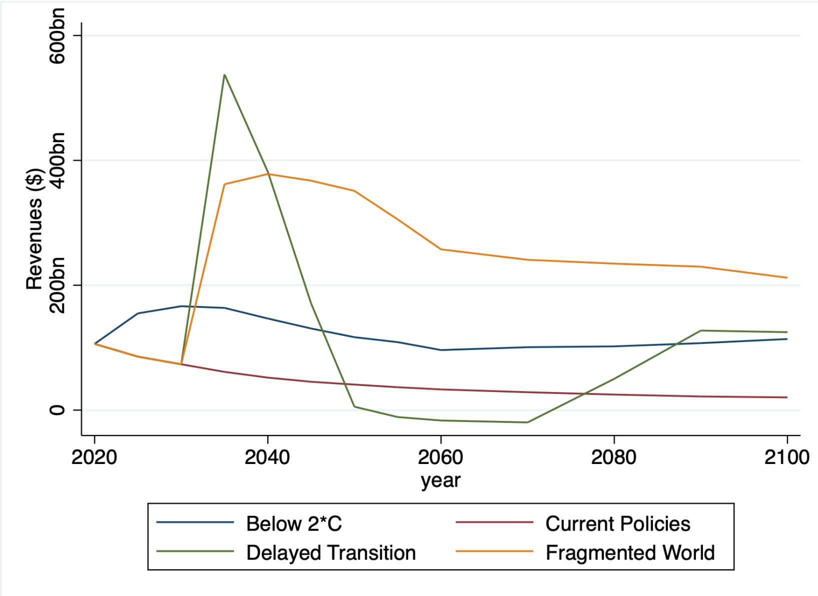


FIGURE 3.7: Carbon Revenues for EU27

In order to determine the resulting EU climate bond issuance capacity, we estimate the present discounted value (PDV) of the revenue starting from 2024. To this aim, we convert the revenues in 2023 US dollars.[22] Next, since NGFS projections are at a five (or ten) years' frequency, we interpolate them to obtain yearly revenues, and discount these real cash flows with the real spot interest rates for the corresponding maturities, as measured by US Treasury Inflation-Protected Securities (TIPS) rates.[23] This enables us to compute the PDV of constant-dollar revenues for each scenario, as of 2024:

$$PDV = \sum_{t=0}^{76} \frac{\text{revenue}_{2024+t}}{(1+r_t)^t}$$

where the estimation horizon ranges from 2024 to 2100 (76 years) and r_t is the maturity-t real spot rate as of 2024.[24] Finally, we convert this constant-dollar PDV into constant-euro PDV

[22]To this purpose, we use the US GDP deflator drawn from the Federal Reserve Economic Data (FRED) by St. Louis Fed.

[23]We draw these data from the Wall Street Journal website for 1-year maturity and from the Federal Reserve's website for longer maturities.

[24]Using this rate, which reflects the currently negligible default risk of US government bonds, is justified if one assumes the probability of default of the EU on these bonds to be equally negligible and the future path of interest rates to be invariant across scenarios. However, the path of default-free interest rates may differ across scenarios. If so, the ETS revenues in each scenario should be discounted by the relevant sequence of interest rates and the PDV should be computed by weighting the discounted revenues in each scenario by the respective probabilities. However, the developers of process-based IAMs explicitly state that probabilities cannot be meaningfully assigned to the climate scenarios they consider: see Dessai and Hulme, 2004 and IPCC, 2007.

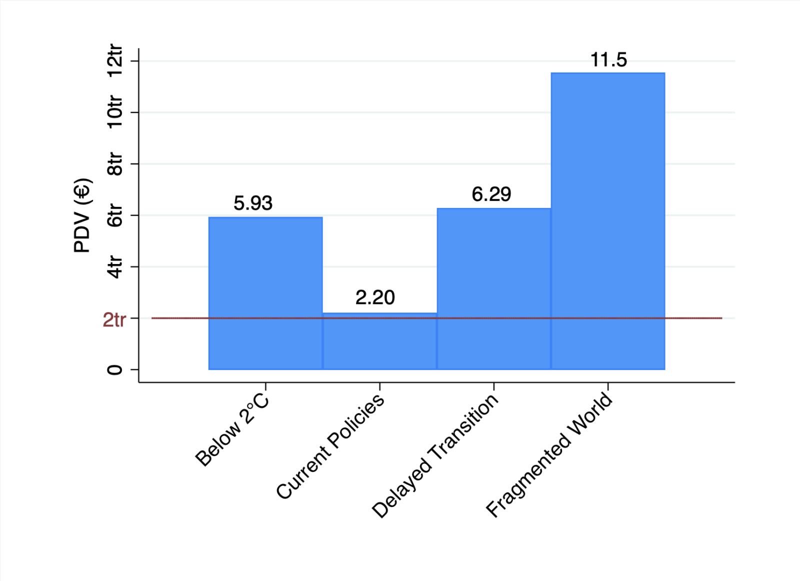


FIGURE 3.8: PDV of Estimated Revenues

amounts, using the euro/dollar exchange rate in 2024 ($1/€0.9167). Figure 3.8 displays the resulting EU climate bond issuance capacity estimates conditional on the four NGFS scenarios.

Figure 3.8 shows that the issuance capacity varies from a lower bound of €2.20tn in the "current policies" scenario to an upper bound of €11.5tn in the "fragmented world" scenario, taking intermediate values close to €6tn in the other two scenarios. However, in all of these cases the issuance capacity of EU climate bonds exceeds the European Commission's long-term budget (6 years) for climate actions (€2tn). It also exceeds the corresponding 6 years climate investment gap, estimated to equal €2.22tn.[25]

3.5 Benefits of the EU climate bond

The estimates presented in the previous section suggest that implementing our proposal would provide the EU with large financial resources to support its climate policies, enabling it to foster the low-carbon transition more swiftly and effectively than with the currently allocated resources. In addition, the proposed reform would have other important benefits in terms of

[25]The estimated 6 years climate investment gap is obtained multiplying by 6 the yearly gap resulting from Section 3.2.

both macroeconomic performance and of capital market development for the EU, which we discuss in this section.

3.5.1 Macroeconomic performance

As already mentioned in Section 3.2, mitigation and adaptation finance initiatives are complementary to tackle climate change. As they both contribute to shield the economy from climate-related losses (e.g. from natural disasters), they also protect countries' fiscal capacity, and enable them to fund climate policies without sacrificing other important policy priorities.

Specifically, spending on climate policies may generate a positive "real feedback loop" via its positive effect on economic growth and fiscal capacity, as illustrated by Figure 3.9. Starting from the top of the figure, faster and larger spending on mitigation and adaptation contributes to increase a country's resilience to climate disasters. Higher resilience, in turn, helps the country to maintain high GDP growth, which strengthens its fiscal capacity, and thus its ability to invest in climate mitigation and adaptation. As a result, spending on climate policies tends to be self-reinforcing and correlate with better macroeconomic performance and higher fiscal capacity.

Figure 3.9 shows that the macroeconomic effects triggered by public climate investments may also generate a "financial feedback loop". By sustaining a country's fiscal capacity, climate policies can contribute to improve investors' expectations about a country's climate risk exposure, lowering its perceived solvency risk. This, in turn, should translate into lower yields on the country's sovereign debt. The resulting lower cost to finance public climate investments would reinforce the positive effect of greater fiscal capacity on climate investing, and thus contribute to increase the country's climate resilience, closing the loop. Thus, the financial loop would reinforce the real feedback loop set in motion by climate policies.

The climate feedback loops illustrated by Figure 3.9 implicitly highlight the potential for multiple equilibria, with climate investment, macroeconomic performance and financial stability correlating across equilibria. If so, the economy may be trapped in an inefficient equilibrium featuring low climate investment and resilience, anaemic growth and high sovereign risk. If inadequate resources are spent on mitigation and adaptation policies, the economy is exposed

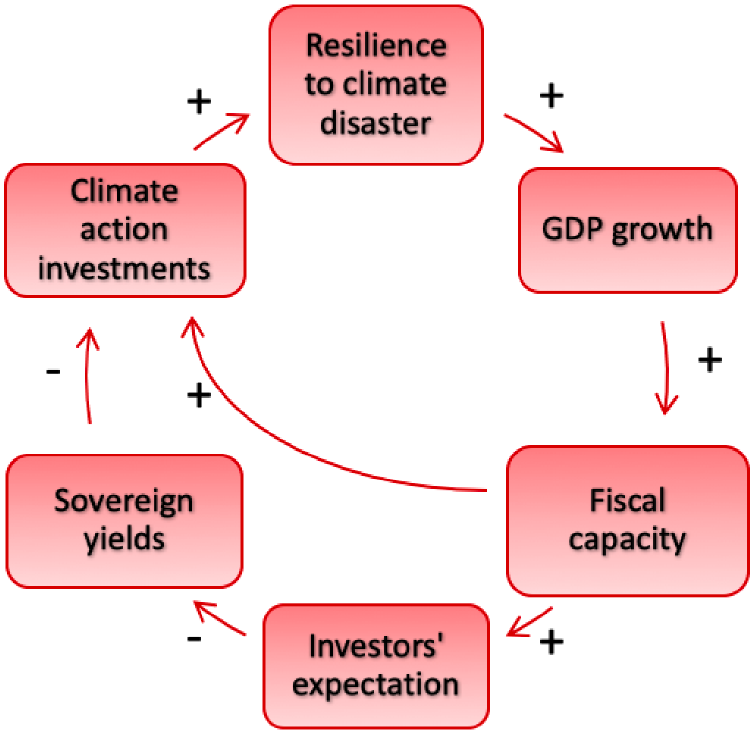


FIGURE 3.9: The real and financial climate feedback loops

to severe and frequent natural disasters, which sap its growth and reduce its fiscal capacity. This, in turn, prevents the government from funding climate investments and policies. The reduction in fiscal capacity increases the likelihood of a sovereign debt crisis, and investors' negative expectations regarding sovereign solvency further hinder the government's ability to fund climate policies.

Mobilizing timely and sufficiently large resources for climate action investments, the issuance of EU climate bonds can avoid such inefficient outcomes, by adding to the resources available to fund climate investments. As illustrated by Figure 3.9, this can set in motion virtuous, self-reinforcing macroeconomic effects, consisting not only of higher and more stable GDP growth, but also of lower risk of sovereign debt crises.

The issuance of EU climate bonds can have an additional, and no less important, benefit in terms of international competitiveness of the European industries catering to the decarbonization process. The funding raised by issuing these bonds can be the financial backbone of the EU response to the ongoing competition from US and China to attract investments instrumental to the low-carbon transition. In particular, it can help Europe fend off the challenge arising from the US Inflation Reduction Act (The White House, 2023), allowing EU MS to attract and support firms that contribute to the decarbonization of the economy.

3.5.2 Safe asset supply and financial stability

The issuance of EU climate bonds can also play a key role in the development and in the stability of European capital markets, by providing a large supply of a safe euro-area asset issued at different maturities.

As already highlighted in Section 3.4.2, EU climate bonds can be expected to be highly liquid, being issued regularly and in large amounts. Investors will also perceive them as a safe sovereign asset, being issued by a supranational financial authority with high credit rating and with the direct backing of the revenue from sales of ETS allowances.

As such, these bonds will be ideally positioned to fill the current demand for a EU safe asset, and address the scarcity in the global supply of safe debt securities.[26] Currently, the euro area does not supply a safe asset to the same extent as the US, although its economy is similar in size and its financial markets are at the same stage of development. Only a few euro-area countries (Germany, the Netherlands and Luxembourg) issue sovereign debt with a triple-A rating by either Moody's or S&P, but their supply of public debt is far smaller than that of the US: in the last quarter of 2022, the face value of central government debt securities issued by these countries amounted to €201bn (i.e. about 1.5% of euro area GDP), while that issued in the US was $1.6tn (6.15% of US GDP).

The scarcity and asymmetric supply of euro-denominated safe assets creates two problems (see Brunnermeier et al., 2017). First, it exposes the European economy to a potential "diabolic loop" between bank risk and sovereign risk, by encouraging banks to be overly exposed to domestic sovereign risk. Second, the asymmetry across countries in the supply of safe assets creates the potential for sudden, self-fulfilling capital flights from high-risk to low-risk countries in search of safety at times of crisis.

However, the financial benefits of the EU climate bonds would not only rest on their safety. By being credibly tied to climate investments, these bonds will represent a new, plentiful supply

[26] This scarcity is witnessed by the fact that the most widely held safe asset, US Treasury bills and bonds, earns a "safe haven" premium of 0.7% per year on average (Krishnamurthy and Vissing-Jorgensen, 2012)).

of a *green* safe asset. Namely, they will be a form of safe sovereign debt that global investors could use to satisfy their growing appetite for environmentally sustainable portfolios. The "greenness" of these bonds would be guaranteed by a system of checks and balances to avoid greenwashing as well as EU MS' moral hazard. To this aim, the use of the revenues obtained from the sale of EU climate bonds would be conditional on their use to fund climate projects in the EU, whose implementation will be monitored via KPIs by the EIB.[27] EU MS that fail to deliver on their KPIs will face a penalty, in the form of reduced allocation of subsequent funding. As such, these bonds are likely to command a greenium relative to comparably safe sovereign assets, such as US treasuries.

3.5.3 Monetary policy conduct

EU climate bonds may also become a key policy instrument for the conduct of monetary policy in the euro area, being a EU-wide safe and green asset. The ECB could employ them to carry out its two main types of monetary policy operations. On the one hand, it could accept EU climate bonds as high-quality collateral in lending to euro-area financial institutions. The haircut rate at which the ECB would accept them as collateral would reflect their safety, sending a strong signal to markets. On the other hand, the ECB could use EU climate bonds as the main asset for open market operations or asset purchase programs. Of course, reliance on these bonds will depend on the extent to which the ECB will maintain a structural portfolio of assets, which is currently under debate within its operational framework review.

Employing EU climate bonds in its operations would have two main advantages for the ECB. First, it would simplify the implementation of its monetary policy programs. When necessary, the ECB could simply decide the size of the programs, without having to discuss the cross-country composition of the assets to be purchased, as well as the distribution of any gains/losses on these assets: the availability of this supranational bond would overcome all concerns about monetary policy giving preferential treatment to any national issuer.

Second, EU climate bonds would represent a simple vehicle for "greening" monetary policy. This would not be a completely new policy: already with its corporate sector purchase

[27] EIB checking on the implementation of EU-funded projects is not a novelty, being already in place in the context of EU Structural and Cohesion Funds, and of the Recovery and Resilience Facility.

program (CSPP), the ECB has tilted its corporate bond purchases towards issuers with better climate performance, measured on the basis of lower GHG emissions, more ambitious carbon reduction targets and better climate-related disclosures. But the availability of EU climate bonds would enable the ECB to scale up considerably its "green" asset portfolio, while taking lower default risk than it would by purchasing corporate debt issued by low-carbon companies. The"greening" of monetary policy has a firm legal basis in the ECB statute: while its primary mandate is to maintain price stability, its secondary mandate is to support the general economic policies in the EU.[28] This includes helping an orderly transition to a carbon-neutral economy, including the promotion of sustainable finance and the creation of incentives for a greener financial system. The availability of EU climate bonds would provide a way for the ECB to fulfill this aspect of its secondary mandate without jeopardising the price stability objective.

3.6 Conclusion

Europe faces a large climate investment gap. To fill it, in this paper we propose the joint issuance of a EU climate bond, to be serviced by the revenues from the sales of ETS allowances. The proposal envisages the extension of the ETS to all sectors (in line with current plans for an ETS 2) and the calibration of ETS allowances supplied by the EC so as to target a science-based carbon price path. This scheme would not only commit EU policy makers to a future path of carbon prices (contributing to reduce transition risk), but would also enable the EC to tap and manage a federal source of fiscal capacity. The revenues should be used to fund climate investing initiatives designed and enforced at the EU level. We show that this scheme could provide a substantial amount of additional funding to EU climate policies: even in the scenario associated with the lowest possible PDV of future revenues, the fiscal capacity generated by this scheme would be of the same order of magnitude as the funding already budgeted for EU climate policies.

[28] Article 127 (1) of the Treaty on the Functioning of the European Union mandates that, "without prejudice to the objective of price stability, the ESCB shall also support the general economic policies in the EU with a view to contributing to the achievement of the Union's objectives as laid down in Article 3 of the Treaty on European Union". These objectives include balanced economic growth, a highly competitive social market economy aiming at full employment and social progress, and a high level of protection and improvement of the quality of the environment.

Our proposal would contribute to improve the climate resilience of the EU. On one hand, the supra-national design of EU climate policies proposed would avoid the inefficiencies stemming from potential cross-border externalities and spillover effects of state-level climate policies. Their joint funding and enforcement would avoid potential moral hazard issues in the national implementation of climate policies, as the funds raised by the issuance of EU climate bonds would be made available to MS conditional on their performance on the implementation of climate investments. At the same time, the issuance of EU climate bonds would increase the speed and efficiency of EU climate investing, by relieving the fiscal constraints that might otherwise deter it in more vulnerable MS.

The proposed scheme can also be expected to have macroeconomic and financial benefits. It would protect economic growth and stability of the EU from the threats posed by natural disasters, and increase its resilience to sovereign crises. Additionally, the joint issuance of such bonds may benefit the international competitiveness of European industries catering to the decarbonization process, by supporting the European response to the ongoing competition from US and China to attract investments for the low-carbon transition.

Finally, the joint issuance of the EU climate bond would provide a large supply of European safe, liquid and green assets, which would both not only meet investors' demand for these assets and provide the backbone for an integrated European capital market, but also enable the ECB to green its monetary policy operations.

Appendix

Tables

TABLE 3.2: EU-27 carbon taxes

Description of the carbon taxes over different EU countries.

Country	Carbon tax (1)	Carbon tax (2)
Denmark	Fossil Fuels	F-gases
Estonia	CO_2eq	
Finland	Transport fuels	Other fossil fuels
France	CO_2eq	
Ireland	Diesel & petrol	Other fossil fuels
Latvia	CO_2eq	
Luxembourg	Diesel	Other fossil fuels
Netherlands	CO_2eq	Electricity & industry
Poland	CO_2eq	F-gasses
Portugal	CO_2eq	
Spain	CO_2eq	
Sweden	CO_2eq	

Source: World Bank

Chapter 4

Sustainable Families

Carla Guerriero and Antonia Pacelli

Household consumption decisions play a central role in studying consumer behaviour and market choice. Pursuing the objective to fight climate change and limit global warming, informing consumers about the carbon footprint of the food products can be a strategy to reach climate objectives. This work aims to analyse through an experiment in the south of Italy the effect of information on health and environmental characteristics on purchasing choices of individual members and family units. Furthermore, it investigates the decision power of individuals within their households, focusing on children. We find that children respond more to the introduction of the labels compared to their parents. Moreover, the presence of labels do have an impact both on individual and collective choices. The results suggest that environmental and health labels are relevant for the decisions of the family unit.

Keywords: ecolabel, information effect, household consumption, decision power, children.
JEL codes: D10, D12, Q56, Q58.

4.1 Introduction

Climate change is humanity's greatest threat, with profound implications for world peace and stability (United Nations, 2021). The United Nations Conferences of Parties (COP) on Climate Change seeks to make world leaders increasingly committed to tackling climate change. However, climate scientists judge that the resolutions are insufficient to meet the Paris climate agreement goals and keep the global temperature to 2°C (Masood and Tollefson, 2021). The current "top-down" approach, centered on policy interventions and regulations, has not delivered the expected results thus far. Exploring bottom-up approaches to mitigate greenhouse gas (GHG) emissions could emerge as a cost-effective and complementary method to achieve substantial reductions.Promoting households' purchasing choices in favour of sustainable diet can produce significant environmental improvements. The food sector is one of the leading contributors in the current evolution of the climate crisis.

The production, distribution, consumption and disposal of food and beverage products has significant environmental impact. Food alone accounts for about a quarter (24-26%) of the global GHG emissions (Crippa et al., 2021; Ritchie and Roser, 2020; Spaene, Dutzler, and Bjelkengren, 2022) and is the priority cluster for the environmental aspects related to emissions, water and land use. In Europe, the impact of household final consumption accounts by 20-30% of the total environmental impacts of consumable goods (Hertwich et al., 2010), and the extraction of resources or the primary production stage - agriculture, fisheries - has a relevant share of the environmental impact of these products in their entire life-cycle (European Commission, 2011). Dietary habits and consumption choices represent one of the main channels for a sustainable lifestyle. As reported by Spaene, Dutzler, and Bjelkengren, 2022, the carbon footprint of typical diets goes from 1.5 tons per carbon dioxide equivalent (tCO_2eq) per person in the vegan diet to 3.3 tCO_2eq in the meat lover diet. As result, promoting households' purchasing choices in favour of sustainable diet can produce significant environmental improvements (Carlsson-Kanyama, 1998).[1] Indeed, the pivotal role of household consumption decisions is evident in the exploration of consumer behavior and market preferences.

[1]Carbon dioxide equivalent refers to a ton of CO_2 or the equivalent amount of other powerful GHG, nitrous oxide (N_2O) and perfluorocarbons (PFCs).

According to Potter et al., 2021, user-friendly, informative and transparent labels can significantly influence behaviour and stimulate demand for environmentally sustainable products. Ecolabels identify products and services with low environmental impact and enable consumers to make informed decisions based on a product's environmental performance (Schwartz, Loewenstein, and Agüero-Gaete, 2020). There has been a growing literature related to the relationship between individual food choices and environmental issues. Yokessa and Marette, 2019 review the results of studies conducted in different countries about the impact of the introduction of ecolabels on the willingness to pay of individuals. The results of the studies suggest a positive effect of the presence of ecolabels on the outcome variable. Moreover, a survey done in 2009 showed that 72% of EU citizens believe that ecolabels should be mandatory (European Commission, 2009). However, the effect of ecolabels on household purchasing choices has yet to be studied.

The objective of this study is to investigates the impact of labels on food purchasing choices of individuals and households. We address this question conducting an incentivized experiment eliciting preferences related to routine food consumption decisions. Compared to previous studies, this work considers both individual and joint decision at the household level.

The experiment is designed to control for potential confounding channels that may drive food selection: taste, healthiness and least but not last, price (Nguyen, Girgis, and Robinson, 2015); we add to these channels the carbon footprint. Building on the recent literature, this paper addresses a set of research questions, on household members' preferences and intra-households dynamics. In particular, this paper studies: (i) what is the effect of information on household consumption choices? (ii) how different household members react to the labels, and in particular to ecolabels controlling for healthiness, taste and price? And (iii) what decision-making power is wielded by each member of the family, and in particular who is the ultimate decision maker when it comes to sustainable products?

This work presents the results of the pilot study, which aims to provide preliminary answers to these questions, paving the way to the final study study which will be conducted with a large representative sample of families in the Campania region. In our experiment, we select 54 families from a middle-income elementary school in the province of Naples. The size of the school was ideal for the pilot phase: classes of 10-20 children per each year. The study was design in collaboration with the Department of Pediatrics of the University of Naples Federico II. The treatment was randomly assigned to each class, and the family related to the children of that particular class were assigned to the treatment or to the control group. Both groups are shown two virtual supermarket shelves with real prices. They all contain the same products, all of them belonging to the food sector.[2] In the Baseline space (control) the individuals will observe the items with real price only; in the Label space (treatment), items will carry, besides prices, an ecolabel and health labels indicating their carbon footprint and whether they are healthy. The first part of the experiment (described in Experiment Part I, Section 4.3.1) is fully incentivized as we provide a supermarket voucher of 50 euros for each family unit, that needs to be used to buy the items selected during the experiment.[3] In order to disentangle the single effect of taste, price, healthiness and environmental sustainability on food choices, we also conduct the Experiment Part II described in Section 4.3.2, which is a discrete choice experiment done individually and jointly, investigating consumption choice in purchasing an hypothetical snack for the child.

The objectives of this project are to investigate the effect of an ecolabel on household decision making process and intra-household decision dynamics using mainly triads (two parents and one child) in a single family unit. The results suggest that labels have an effect on both individual and collective choice. Children are particularly responsive to labels, and it seems that they influence collective choices. Moreover, among different attributes taste plays a leading role in food choices; however, also health and environmental characteristics are relevant attributes.

We find that the eco and health labels do have an impact on consumption choices at household and individual level. In particular, children react more to the information treatment compared to their parents. Fathers seem to be more willing to choose more sustainable and healthy products on average, but the reaction to the labels is similar to the mothers' group. A reason could be that, in this sample, it is generally not on the fathers' duty the grocery. Therefore, mothers are more close to their actual daily choices. The second part of experiment conducted in the study investigates individual and family choice in a DCE on child daily snack. The results

[2] Items were selected are the most consumed by Italian families following ISTAT reports.

[3] We made an agreement with the selected supermarket, which provided us the bar code for each voucher in order to observe the items bought by the family unit. The participants were fully informed about this procedure through the inform consent.

of the second experiment suggest that taste plays a leading role when individuals and family are asked to decide the food to buy, followed by health and environmental concerns. Price displays the expected sign (negative) only in the parents sample but is not statistically significant indicating that, given the price range considered, it is secondary to the aforementioned attributes.

The paper is organized as follows. Section 4.2 reviews the literature and places the paper in the recent research scenario. Section 4.3 describes the design of the experiment in its two different parts. Section 4.4 illustrates the data and the labels utilized in the study, followed by Section 4.5 which describes the methodological analyses performed. Then, the results are shown in Section 4.6 and Section 4.7 concludes.

4.2 Literature Review

This paper relates to two strands of literature: the effects of labels on individual consumption choices (environmental and health labels) and household collective model.

Diverse disciplines emphasise the importance of food purchasing decisions on health, and it in some points intercepts with climate concerns (Parodi, 2018). In recent decades there has been a progressive change in eating habits, with a significant increase in the consumption of ultra-processed foods rich in sugars and saturated fats. This change has gone hand in hand with a reduction in physical activity resulting in an alarming increase in obesity rates since early childhood. The aforementioned consumption patterns pose a threat not only to public health but also to the "well-being" of our planet, considering that agriculture and food production processes are responsible for releasing around 25% of all greenhouse gases (Willett, 2019). Therefore, establishing healthy and sustainable eating habits from early childhood can help prevent obesity and related diseases and simultaneously reduce nutrition's environmental impact, given that childhood food choices tend to persist even in adulthood (Birch, Savage, and Ventura, 2007). Children's eating behaviour is influenced by parents, who actively choose foods and serve as role models for food choices; at the same time, parenting practices are also influenced by the characteristics of the child, such as age, sex, weight and eating behaviour (Birch, Savage, and Ventura, 2007). As demonstrated in a recent study conducted in Belgium on parents of children between 6 and 12, socio-economic factors can impact whether or not to choose healthy and sustainable food (Vos et al., 2022). Furthermore, cultural factors can also influence nutritional choices, as demonstrated in a study of Norwegian adolescents (Fismen, Samdal, and Torsheim, n.d.). Nutritional labels distinguish a healthy product from a less healthy one. However, basic nutrition knowledge is required to interpret nutritional labels and the availability of time to compare the nutritional parameters of different products. This can make choices more difficult for low-income or lower-educated families. As demonstrated in a study conducted on low-income families, front-of-pack (FOP) nutrition labels can be a valuable support for consumers, providing quickly accessible and understandable nutritional information. Furthermore, "plain" FOPs are more helpful in selecting healthy products than FOPs that present nutritional information (Blitstein, Guthrie, and Rains, 2020).

Looking at previus works investigating factors affecting consumers' "green" purchasing behaviour it emerges that the ecolabel may have the potential to change behaviour and stimulate the demand for more environmentally sustainable products (Joshi and Rahman, 2021), but the effect of ecolabels on individual household members' and families' collective consumption choices has not yet been investigated (Rokka and Uusitalo, 2008).
The household, or family, has long been recognized as a key decision-making and consumption unit. From the economic perspective, the family originates as a partnership between two individuals who decide to jointly produce and consume a broad range of goods and services, including affection and children (Bruyneel et al., 2017). Based on neoclassical utility theory, the Unitary Model of household decision making assumes that choices are based on a single household utility function, which is maximized subject to a pooled household budget constraint (Browning, Chiappori, and Weiss, 2014). Despite the advantages of this model, such as testable restrictions on household behaviour (e.g. symmetry of the Slutsky matrix and the income pooling property), the theoretical and empirical evidence has rejected the hypothesis that a household composed of two (or more) individuals can behave like a single economic agent maximizing a single utility function under a single budget constraint. Given the mounting evidence against the assumptions of the Unitary Model, most researchers now agree that collective household models allowing for differing spouses' preferences and the intrahousehold distribution of decision-making power offer a more realistic description of the economic

behaviour of a two-member household (Browning, Chiappori, and Weiss, 2014).

Throughout the long history of academic study of the family, economists have paid almost no attention to the possible influence of children on household decisions. Economists have provided less and less space to the importance of families in recent years and, in particular, to the role of children, who directly influence parents' consumption decisions List, Petrie, and Samek, 2021. The studies focus mainly on the effects of the choices, but not on the path taken to reach them Commuri and Gentry, 2000. Children have an increasing economic impact as consumers Bruyneel et al., 2017. Most of the literature is focused on their joint decisions with parents and peers (Calvert, 2008, Wouters et al., 2008, Andreoni, Gillen, and Hardbaugh, 2002, Cox, Friedman, and Sadiraj, 2008), which is a crucial aspect of the decision making process of children.In the household behavioural model, young children are often considered as mere spectators to parents' actions. Even where collective models are used in lieu of unitary models, accounting for individual preferences, the role of children is generally incorporated in the model in the form of parental "caring preferences" or else in respect of the public goods that children consume (Browning, Chiappori, and Weiss, 2014). Economists justify this exclusion in two ways: children lack preferences for their own consumption, and they also lack the financial autonomy to buy what they prefer (Dauphin and El Lahga, 2011). However, according to Calvert, 2008, children do influence family buying decisions over a broad range of goods and services, not only meals but even such big-ticket items as cars and vacations. Empirical studies have also found that even young children regularly receive money from their parents; they earn money; and, especially in low-income countries, they contribute significantly to household income (Furnham and Gunter, 2008; Furnham, 2008).

Preliminary evidence suggests that children have definite preferences for environmental protection and that they influence their parents' environmentally relevant behaviour. Dardanoni and Guerriero, 2021 showed that children are willing to give up part of their own money in order to protect the environment. A study by Dupont, 2004 showed that the presence of children influences households' willingness to pay for environmentally better goods. Lawson et al., 2019b provides evidence that upward inter-generational learning – the transfer of knowledge, attitudes or patterns of behaviour from children to parents – can be a powerful pathway through which children prompt concern over climate change among their parents.

The age at which children begin to influence household decisions for sustainability as decision makers themselves or indirectly through parents' caring preferences is still an open question. Despite the progress made in accounting for the complexities of two-member-household decision-making, little is known about the process when children are present or the extent to which their preferences may direct household choices towards environmentally sustainable products.

Little is known about the decision power of children within the household decision-making process. Nowadays, this lack is highlighted due to climate change: children of our historical era are sensitive to environmental issues and engines of change. Present and future generations may bear the disproportionate costs of mitigating the effects of climate change (The Future of Children, Princeton, 2016). Clear examples are the international school strikes promoted by *FridaysForFuture*, which fights for sustainability to guarantee the future to young generations. Therefore, the growing sensitivity to the topic of children role is directly connected with the influence that children might have on the consumption choices of their families (Lawson et al., 2019a). However, there is little empirical evidence on the role played by children in influencing their parents' "sustainable" consumption choices (Dauphin and El Lahga, 2011). Focusing on a particular household member, to the best of our knowledge, very few studies have elicited children's preferences and compared them with their parents' in stated-choice experiments. Guerriero et al., 2017 assessed whether children's willingness to pay for a reduction in health risk was influenced by their parents' willingness. The study found a significant positive relationship between parents' and children's willingness to pay, even controlling for other demographic variables (e.g. age and gender of the child). A second study on the same parent-child dyads investigated children's willingness to pay for environmental protection using a discrete choice experiment (Dardanoni and Guerriero, 2021), concluding that a higher degree of environmental concern in parents decreases the probability of children's choosing the "No Environmental Action" alternative. Families with children may be more environmentally sensitive. Young people are not only victims of climate change; they are also highly sensitive to environmental issues and, as *FridaysForFuture* demonstrates, they can be powerful agents of change. But there is only limited evidence concerning the power of children in households' decision-making.

4.3 Experimental Design

The experiment is conducted at school from the 13th to the 17th of November, 2023. The first meeting is in the classroom, where a short interactive group discussion takes place about the importance of scientific research and the steps of the project. Then, the pure part of the experiment stars and it consists of a meeting where families were invited to the school to do the questionnaires. To avoid members of the same family can influence each other, they were asked to be separated and choose individually. Then, the household unit completed another survey choosing all together the answers to the questions proposed. In the end, all children enrolled with parental support filled in a validated food diary to identify eating habits, the caloric intake of macro and micronutrients.

The study consists of two part of the experiment: part I, through the randomized controlled trial (RCT) we analyse whether individuals and families choose greener and healthier alternatives in the presence of information on the environmental and healthy impact of the goods. The incentive to participate for the families is a supermarket voucher for grocery. Part II, through a discrete choice experiment, the authors explore the role of different attributes (health, environmental sustainability, taste and price) on the food purchasing choices of individual members and family units.

4.3.1 Experiment Part I - Randomized Controlled Trial (RCT)

The RCT is an experimental method for evaluating policies trying to understand the causal effects of the introduction of policy tools with a high level of confidence. The agents are assigned completely random to the control or the treatment group. The latter receives the information treatment object of the study, while the control group does not, since it represents the counterfactual. In this experiment, the counterfactual situation is the observation of the product and the price, as it happens in ordinary supermarkets. The choice that the families randomly assigned in the control group make, it is the situation without the information treatment, therefore the counterfactual situation. For being the counterfactual of the treatment group, the underlying assumption is that the participants have similar characteristics, and the outcomes of the experiment are associated with the treatment effect rather than individual characteristics. This is true for this study, since the socio-economic status and characteristics of families belonging to the same school are homogeneous, as shown in the descriptive statistics in Table 4.1. In our sample, we have 9 single-mothers, 8 with middle-school title and 1 with high school diploma.[4]

Families					
Education					
Family level	Freq.	Percent	**Parents**	Freq.	Percent
			mothers		
both middle school	13	24.07	middle school	15	27.78
middle-high school	13	24.07	high school	30	55.56
both high school	17	31.48	university degree	9	16.67
high-degree	10	18.52	**fathers**		
both university degree	1	1.85	middle school	16	35.56
			high school	26	57.78
			university degree	3	6.67
Single-mother family	9	16.67			
Occupational status					
Family level	Freq.	Percent	**Parents**	Freq.	Percent
			mothers		
both unemployed	16	29.63	unemployed	36	66.67
unemployed-employee	16	29.63	employee	17	31.48
both employees	19	35.19	self-employed	1	1.85
employee-self employed	2	3.7	**fathers**		
both self-employed	1	1.85	unemployed	10	22.22
			employee	29	64.44
			self-employed	6	13.33
N of families	**54**				

TABLE 4.1: Description of family units in the sample.

Since the attention on sustainability and healthy food has exponentially increased over the last decades, it is reasonable that people on average consume more green and healthy food, if they are informed about it. However, consumers still tend to choose non-green alternatives, probably due to the existence of some trade-offs that might prevail (price, brand, trust). Moreover, thinking about daily products (e.g., pasta, oil, in the case of Italian families) people might be inelastic on their preferences, and hardly decide to change their usual alternative with a different one.

As shown in the following section, the ecolabel and health label used in this study follows the colours and the structure of the nutrition score NutriScore. Labels are designed to be easily

[4] In four cases, single-mothers live with the grandmother.

understandable also for children from age 5. Indeed, this pilot study is a test to ensure satisfactory understanding and scenario acceptance by respondents of all ages (Elliott, 2009). The ecolabel is expected to inform consumers about GHG emissions using a five level scale, from 1 to 5 trees. The health label informs on the health characteristics of the product, from 1 to 5 stylized men.

In our experiment respondents face two different scenarios:

1. Baseline: status quo; people in this group are informed only of the price of the various items;

2. Label: this group has access to information on the carbon footprint and the health impact of each item and the relative price.

The children, parents and the household as a unit answered to the questionnaires. Each household was randomly assigned to the control group (C) or to the treatment group (T). The control group follows the Baseline structure, while the treated families follow the Label scenario. The difference between these two groups is that the treatment group observes the eco and health labels in the choice of the goods. Therefore, in the first part of the questionnaire individuals (and then family collectively) had to choose for each food category (pasta, eggs, oil, meat, fish, snacks, milk) they had to select among five alternatives the preferred one.The categories of goods selected for the questions of the experiment are the most consumed by Italian families following ISTAT reports.[5] The control group chose based on the product and the price, like in a real supermarket shelve. The treated group observed also our labels on health and environmental characteristics. An example of a question of the experiment is presented in Figure 4.1 and 4.2. The prices of the goods are the real prices of the supermarket.

The authors were able to collect two kinds of data: i) the preferences of individual household members and ii) household decisions as a unit. The experiment allows all the respondents to make choices autonomously without the control of interviewers, as a way of mitigating social desirability bias. The experiment is incentivized: a fixed budget was assigned to each household (50€) in form of voucher at the near supermarket[6]. They are asked to buy the item

[5] See ISTAT website.

[6] In the pilot case, the Sole365 supermarket chain.

FIGURE 4.1: Example of the labelled question of the experiment.

FIGURE 4.2: Example of the unlabelled question of the experiment.

resulting from the choices made by one household member or by the household collectively, selected at random. This means that all participants have the same incentives during the experiment. We also collect information on socio-demographic characteristics as well as individual beliefs and attitudes/action on climate change (Andre et al., 2021).

First of all, this work aims to answer the questions related to the effects on consumption choices of a labelling system on the environmental and health impact of products, analysing potential inter-generational differences, which we describe here. In the next paragraph, it is described the second part of the analysis where we start to study and explore the role of the child in the consumption choices of the family unit.

4.3.2 Experiment Part II - Discrete Choice Experiment (DCE)

A discrete choice experiment (DCE) is a frequently used research method in economics for eliciting consumers' revealed preferences, and it is particularly common to evaluate public goods. In a DCE, researchers show several hypothetical choice scenarios to the participants, called choice sets. These scenarios involve a number of alternatives, described as a set of mutually exclusive attributes, from which participants must choose their preferred option. The attributes reflect different characteristics of the good: that way preferences are revealed without participants being explicitly asked to state their preferred level for each attribute.

The DCE proposed in this study was designed and performed so as to analyze how individuals in families and the family as unit of analysis make their choices when selecting their favorite food item. In particular we aim at disentangle the effect of four attributes in the chioice process. After participants were recruited, they were asked to fill in a survey and each member of the family (mother, father, child), and then collectively, had to submit their preferences through it.

The attributes presented in the DCE are: price, taste, environmental and health characteristics (using the same labels proposed in the first part to the treated group).

The DCE part of the survey contained seven choice sets, generated following the method proposed by Hole in its work on the optimal design of discrete choice experiments (Hole, 2015). An example of the questions proposed is shown in Figure 4.3. The participant was ask to imagine to choose a snack for the child (and the children for themselves), knowing that each alternative would have the described characteristics.

Each choice set consists of four alternatives. Since that we want to estimate the weight that children, parents and family as a unit give to environmental and health characteristics, price

Caratteristiche	A	B	C	D
Ambiente				.
Salute				.
Gusto				.
Prezzo	7,50€	5€	2€	.
Quale alternativa preferisci?				

FIGURE 4.3: DCE choice set.

and taste, these are the attributes of each alternative. Every choice set contains the status quo or opt-out option (D), the opportunity to do not choose any of the alternatives proposed. The participants had the legend available explaining the labels and the scales of each attribute.

After the second part of the experiment, respondents were presented with a questionnaire eliciting through multiple question their environmental attitudes, personality traits (different questions were used for parents and for children). Raven Matrices were also used to test individual IQ.

In the methodology section we describe how we analysed the outcome of the two parts of the experiment.

4.4 Data

This section illustrates the structure of the surveys and the data collection procedure. The database is originally collected from face-to-face questionnaires. The study recruited households from an elementary school in the Metropolitan City of Naples (in the fraction of Agnano), in November 2023. Naples is the third largest city in Italy and one of the most densely populated cities in Europe. The city is also characterized by wide income and cultural differences, and it offers a good setting for collecting representative sample data on the population. The sample comprises 54 families, consisting of 103 parents and 84 children, totaling 187 individuals (n=241 surveys). The school's size was suitable for conducting the study and piloting the questions, enabling us to gather preliminary insights and results through the questionnaire.

4.4.1 Labels

Necessary for this study is to obtain a reliable indicator of environmental and health characteristics of the selected good. For these purposes, we rely on the collaboration with EcoStep Foundation for the environmental score and with professors of the Department of Pediatrics of the University of Naples Federico II for the health score.

The ecolabel: EcoStep

The EU eco-labelling scheme aims to promote goods with limited environmental impact in their entire life cycle and inform consumers about the effect of the products that they purchase (Stara Zgora Regional Economic Development Agency, 2020). There are thus several benefits coming from the introduction of the eco-labels: consumers acquire knowledge on what they consume; public authorities might faster implement environmental policies and introduce eco-friendly and sustainable criteria in public tenders; also, it creates a new branch of marketing opportunity on green, eco-friendly and sustainable products and services. Currently, EU Eco-label covers a wide range of product groups, like personal care products, clothing and textile products, tourist accommodation and others; nevertheless, it does not include food, feed and drink products. The feasibility report of the European Commission, 2011 shows that they decided to do not to develop Eco-labels related to these categories at that time. The reasons that led to this choice were diverse: first, the impact of these products was not easily measured, and it might make it hard to include them in a general ranking of environmental effects. The same consideration was done for ethical and social issues (animal welfare, labour standards and fair producer prices), which interact with environmental issues. They should be covered too by the Eco-labels. Therefore, the criteria and the compliance are very complex and resource-intensive for the institutions. In the study, they did a consumer survey which provided a lot of helpful information on the perception of the eco-labels for food, feed and drinks. They impact purchasing decisions, but most participants declared to feel confused because EU Ecolabel could overlap with Bio and organically produced certificates and labels related to nutritional features. A solid, informative campaign must inform citizens and make them aware of the meaning of different labels.

Nevertheless, in Europe, there exist lots of labels in the food and beverage industry that provide a wide range of information related to the product's characteristics. For giving some examples, AENOR label has as aim to recognise environmental friendly products and services through some criteria - among others, carbon footprint, zero waste, energy management, safety in the food chain and containers. Other examples of labels in the food industry that captured our attention are Carbon Reduction Label, Climatop, Environmental Product Declaration, FairWild and FairTrade.[7]

Focusing on the Italian context, Altroconsumo recently surveyed consumers on the use of ecolabels in Italy; almost 50% of the respondents said they had green purchasing habits, but just 33% thought they could identify accurate, trustworthy eco-friendly labels and wanted to get standardized and readily comprehensible information on the environmental impact of the goods purchased.

It is generally true that most of the existent labels intercept environmental and social criteria with nutritional aspects, that in our opinion, should be treated separately. The reason that leads us to this idea is motivated by the existence of healthy but not sustainable products and vice-versa. This aspect generates confusion in consumers' perception, which has to separate healthy and sustainable elements.

The partnership with EcoStep Foundation generated significant value to the *Sustainable Families* project.[8] EcoStep Foundation aims at promoted and support the circular economy. Their key focus is the advancement of digital product identification and product tracking, recognizing the transformative potential of technology in fostering environmental stewardship. They propose a globally unique product identity based on cradle-to-cradle accountability, assigning an ID to every piece of product. This system includes a tamper-proof product declaration, ensuring that subsequent modifications and certifications can be transparently tracked and safeguarded against manipulation. Moreover, their application computes the ecological impact of

[7] website of all ecolabels in Europe: see here.

[8] The collaboration consisted in providing us the *EcoStep score* for each selected product in our experiment, along with the related graphics. The same graphic was applied for the *Health Score* used in the study.

products at Scope 3 level. Their dataset aims to be openly and freely accessible. Their dataset is designed to be openly and freely accessible, emphasizing their commitment to traceability and transparency. In addition to these initiatives, EcoStep provides comprehensive information for food products, including ingredient lists and the corresponding *Nutriscore* ratings, alongside the Ecological Impact Score based on life cycle assessments, which is available for all the categories of products, called (*EcoStep Score*). [9]

Health Label

The health label applied to the products in the proposed study are based on the assessment made by our partners of the Department of Pediatrics. Their estimates are based on the principles of the Mediterranean Diet, which took into account the Mediterranean diet food pyramid. It advises to have abundant consumption (several times a day) of bread, pasta (preferably whole grain), vegetables, legumes, fruits, and nuts, moderate consumption of fish, white meat, dairy, and eggs. Laslty, it very limited consumption of red meat. The Health Score does not differ significantly from the Nutriscore. However, the latter is not available for many of the items proposed: the Nutriscore does not apply to unprocessed products that consist of a single ingredient or a single category of ingredients (e.g., fresh fruits or vegetables, raw cut meats, honey). This limitation lead us to propose a new indicator, based on scientific knowledge.

The labels are shown in Figure 4.4 and Figure 4.5. They follow the ratio and the colours of the Nutriscore. However, we chose to avoid letters and use number and elements to be easier to understand for children from age 5.

FIGURE 4.4: EcoStep score, our ecolabel.

[9] The EcoStep Platform stores data in three different storage locations: the distributed ledger (DLT) for all data that needs tracking, a distributed data store for aggregated statistical data, and a distributed file store for uploaded documents such as certificates. It pulls additional data for product scoring from LCA (Life Cycle Assessment) databases and other relevant sources as needed. The Platform has a set of public interfaces to allow external clients to read data anonymously. And a set of private interfaces to create and update product data by organisations and users that are registered with the system. Any sort of client (from desktop/web/mobile applications to sensors) can use the APIs. EcoStep has an own user interface (UI) to visualise products and aggregation entities (like locations, organisations and individuals).

FIGURE 4.5: Health score, our health label.

4.5 Methodology

In order to analyse the first part of the experiment, for assessing the impact of the eco and health labels on the purchasing choices of individuals and households, we estimate the following regression:

$$y_{mean,i} = \alpha + \beta T_i + \epsilon_i \tag{4.1}$$

where y is the mean for each group of the label score, both for the eco label and health label. The analysis is performed at individual level (parents and children) and collectively.[10]

In the second part of the experiment, respondents (individuals and families) were presented the same choice sets (in the same order). The first six sets were built using the experimental design of the DCE was generated using the Ngene software in accordance with the principle of D-Efficiency.[11] The seventh-choice set was designed to test whether the respondents understood the DCE by including a dominant choice option. DCE estimation relies on the random utility model which assumes that the individual choice probability p_{ij} of an individual i chooses alternative j equals the likelihood that the utility of the alternative j is bigger or equal to the utility associated with the alternative q for each alternative in the choice set (q=1, 2..J) (McFadden et al., 1974). Each individual is presented with 7 choice sets (T=7).

$$p_{ij} = prob[(V_{ijt} + \varepsilon_{ijt}) \geq (V_{iqt} + \varepsilon_{iqt}) \forall q \in q = 1 \ldots J; j \neq q] \tag{4.2}$$

where V_{ijt} and ε_{ijt} are the observed and unobserved components of individual/collective utility associated with the alternative j in the choice set t. V_{ijt} can also be expressed as a vector

[10] The results are robust if we add as control gender for individuals and age for children.

[11] The coefficient estimates obtained in the present study will be used as priors to generate the final survey design. Compared to orthogonal designs that are often used in DCE, D-efficient design allows to satisfy several proprieties: level balance, moderate attribute level overlap.

of coefficients β' multiplied by X_{ijt} that are the attribute levels of the attributes of choice alternative j with coefficients β. The error component ε_{ijt} is assumed to follow a type one extreme value distribution. A Mixed Logit model incorporating normally distributed parameters is employed to estimate coefficients, taking into account the diversity of taste among respondents. Another significant advantage of adopting the Mixed Logit model, in contrast to the standard conditional logit, is its avoidance of the restrictive independence of irrelevant alternatives property associated with the logit model. Furthermore, considering that the same respondent is tasked with responding to seven choice sets, the Mixed Logit accounts for unobserved factors that may persist across multiple choice sets for a given respondent. This feature enhances the model's ability to capture patterns in decision-making that may be influenced by factors not explicitly observed in the data.

With specific reference to the present study individual utility can also be formalized as follows:

$$U_{ijt} = \delta_i ASC_{sq} + \alpha_i Price_{jt} + \beta_{li} Env_{jt} + \beta_{wj} Health_{jt} + \beta_{ci} Taste_{jt} + \epsilon_{ijt} \tag{4.3}$$

Where ASC_{sq} is the alternative specific constant for the status quo taking value of 1 for the alternative describing the buy-nothing scenario (D). Price, Env, Taste and Health denote the price level, the sustainability, taste and healthiness levels respectively. To investigate whether the age of the child affect the utility of the four attributes (healthiness, sustainability, taste and price) we interacted the child class attended (from one to five elementary grade) with the attribute coefficients. Results are reported for parents and children separately in the appendix.

4.6 Findings

The sample, due to the size and the characteristics, is not fully representative of the population of the city of Naples. However, it was important for a pilot phase to reach a school with an heterogeneous environment and a reasonable size. The sample is pretty balanced across the two groups (treated and control), allowing the analysis to be reasonably meaningful.

4.6.1 Results Part I (RCT)

The randomized controlled trial offers an interesting perspective on the effects of information on purchasing choices, through label introduction. Tables 4.2 and 4.3 provide a description of the sample and its composition. The summary statistics are presented on the whole sample and then between control (C) and treated groups (T). The sample exhibits a balanced representation between male and female participants. The fraction of unemployed in this sample is large. This is particularly driven from female, since the 69% of them declared to be housewives. From the level of education of the parents and their work placement, the school is classified as medium-low socioeconomic class, which is in line with the neighborhood that hosts the school, where the pilot was conducted. We also collected data for the level of intelligence quotient (IQ), using eight verified questions from the Raven tests. Among the parents, the mothers on average have a slightly lower grade, except for the treated group. In the children sample, females on average perform better, both in the treated and control groups.[12] Then, we have information about their degree of environmental attention: children pay on average more attention to the environment, and females more than males. In the parents sample, as for the IQ, on average fathers declare to be more sensitive to the environmental topics, with exception of the mothers' treated group. The families were assigned to the treated or control group following the allocation of the children.

The results of the first part of the experiment are shown in the Table 4.4 and Table 4.5. In all the groups considered (parents, children and family as a unit) both the eco and health label had an effect on the average choice of the treated agents. Indeed, they choose greener and healthier alternatives compared to the control group. These results is in line with the literature of eco label introduction (Lohmann et al., 2022; Muller, Lacroix, and Ruffieux, 2019). However, the magnitude of the result is sizeable and the inclusion of children suggests that they are more responsive to label introduction compared to their parents. Indeed, we observe that parents choose greener alternatives for 6.14% more in the treated group compared to the control group ones. The average response in children doubles the previous category: 15.34% greener alternatives. When they decide collectively, their change towards eco-friendly products is about 10.3%. This intuitively suggests that children have a role in the collective choice, leading the

[12] The range of the IQ level goes from 0 to 8.

Parents		Total	Control	Treated
Gender (%)	male	43.7	18.45	25.24
	female	56.3	22.33	33.98
Educ (%)	elementary	1.94	0	1.94
	middle school	31.07	13.6	17.5
	high school	55.34	23.3	32.04
	degree	11.65	3.88	7.77
Work (%)	unemployed	48.54	17.48	31.07
	employee	44.66	23.30	21.36
	freelance	6.80	0	6.80
Env attention (level)	male	4.067	4.158	4
	female	4.056	4	4.13
IQ (level)	male	4.6	4.63	4.58
	female	4.54	4.18	4.78
N		103	42	61

TABLE 4.2: Description of the parents in the sample.

parents to choose more eco item.

The same mechanism applies to the health results: children are more responsive than the parents (7.64% vs 4.56%) and the choices of the family as a unit falls in the middle of the two (6.74%), but closer to children percentage of change towards healthier alternatives. The coefficients are all statistically significantly different from zero.

HNevertheless, disentangling the impact of eco and health labels proves challenging, as these attributes are frequently correlated in food items. The second phase of the experiment comes into play to address this issue, aiming to define the role of each attribute within distinct groups.

4.6.2 Results Part II (DCE)

Table 4.6 presents the results of the DCE analysis with Mixed Logit Model for collective choices, parents and children respectively. All attribute coefficients, with the exception of the price coefficient, are positive and significant at the 1% level in all the three samples considered. The price attribute exhibits a negative coefficient solely in the parents' sample, though it is

Children		Total	Control	Treated
Gender (%)	male	53.6	22.6	30.95
	female	46.4	20.24	26.2
Env attention (level)	male	4.2	4.368	4.529
	female	4.667	4.0769	4.773
IQ (level)	male	1.977	2	1.961
	female	2.307	2.117	2.454
N		84	36	48

TABLE 4.3: Description of the children in the sample.

VARIABLES	(Parents) eco_mean	(Children) eco_mean	(Collective) eco_mean
T	0.307***	0.767***	0.517***
	(0.102)	(0.130)	(0.136)
Constant	2.850***	2.659***	2.747***
	(0.0782)	(0.0986)	(0.104)
Observations	103	84	54
R-squared	0.083	0.296	0.218

Standard errors in parentheses *** p<0.01, ** p<0.05, * p<0.1

TABLE 4.4: ecolabel effect.

VARIABLES	(Parents) health_mean	(Children) health_mean	(Collective) health_mean
T	0.228***	0.382***	0.337***
	(0.0701)	(0.0781)	(0.0985)
Constant	3.803***	3.460***	3.636***
	(0.0539)	(0.0591)	(0.0758)
Observations	103	84	54
R-squared	0.095	0.226	0.184

Standard errors in parentheses *** p<0.01, ** p<0.05, * p<0.1

TABLE 4.5: health label effect.

not statistically significant indicating that choices are not sensitive to the prices which become secondary when other attributes are involved in the selection process. The positive coefficients for health, taste, and environment align with expectations, indicating that higher tastiness, healthiness, and a lower carbon footprint are associated with a higher degree of utility for respondents when making food purchasing choices. Specifically, in the parents' sample, the choice of child snacks is primarily driven by taste, followed by healthiness, with environmental sustainability ranking third, albeit less prominently. Children and collective choices follow the same attribute ranking, however, the size of the taste coefficient is larger in the kids' sample. In addition, for the children sample, there is no significant difference between the coefficients for health and environment.

The status quo, namely the buy nothing scenario, is positive and significant only in the children sample. This result is consistent with previous literature indicating that the choice of the status quo is a response to the perceived complexity of the DCE. Increased complexity can indeed generate a desire in the younger respondents to retain or to avoid not to choose, also known as omission bias (Burton and Rigby, 2012). The standard deviation of the parameters of the attributes: taste, environment and health in the parents' sample is high and significant indicating a substantial variation in preferences. The standard deviation is higher in the collective choice for taste and environment while it is not significant for health. Kids preferences seem to be more homogeneous with respect to the attributes considered, because the standard deviations are significant at 10% level for taste and 5% level for environment.

Results of the Mixed Logit analysis for mothers and fathers (see Appendix 4.7), suggest that fathers provide more attention to the environmental attribute compared to the mothers. Mothers predilige the taste attribute in the selection of the snack for the child, while the fathers focus more on the healthiness and the environmental aspect. Studies comparing food choices of mothers and their children found that children are more paternalistic than their mothers when choosing a healthy snack for somebody else, in the study of the mothers (Marette et al., 2016). In our experiment children and parents are asked to choose individually a snack for the child, and the health coefficient is not different between mothers and children. Additional insights into the heterogeneity in individual preferences could be potentially gained by estimating interactions between choice attributes and individual characteristics such as IQ, and the degree of environmental attention. However, in our case the estimated coefficients are not significant (see Appendix 4.7.1). Due to the sample size, these results are only preliminary and should be interpreted with caution.

	(Parents) choice	(Kids) choice	(Collective) choice
Mean			
price	-0.00391	0.0182	0.0144
	(-0.13)	(0.75)	(0.44)
health	0.374***	0.335***	0.427***
	(6.93)	(6.72)	(6.30)
taste	0.432***	0.564***	0.510***
	(9.27)	(11.14)	(6.17)
environment	0.247***	0.308***	0.328***
	(5.43)	(5.77)	(4.17)
status_quo	0.909	2.521***	1.172
	(1.93)	(5.53)	(1.08)
SD			
price	0.191***	0.0616	0.0794
	(5.17)	(1.10)	(1.20)
health	0.305***	0.0629	0.132
	(5.13)	(0.39)	(0.84)
taste	0.238***	0.165*	0.360***
	(4.30)	(2.56)	(4.38)
environment	-0.160	0.194**	0.335***
	(-1.79)	(2.63)	(3.58)
status_quo	2.218***	1.725***	3.088**
	(4.26)	(4.86)	(3.23)
N	2884	2352	1512

t statistics in parentheses
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

TABLE 4.6: Mixed logit results

4.7 Conclusions

This study shows the result of a large pilot study performed in a school in the metropolitan city of Naples to explore the effects of the introduction of the eco and health labels on individual and family food consumption choice and intra-household choice dynamics. The objective of this work is to examine several unexplored research questions about the effect of labels on households' food purchasing choice. First, even if exists a literature on the effects of health label, we know little about eco label effects. More importantly, anadvantage of this study is that we can rely on an ecolabel on food which is transparent. We provide a first evidence of eco and health labels effect on both parents and children consumption choice.

As can be seen from the European Green Deal strategy, the issue of consumer awareness of their purchasing choices is of profound importance for the structure of the regulations that will guide us in the coming decades. In light of this, the European Commission will propose a mandatory harmonised nutrition labelling and develop a framework for labelling sustainable food products from an environmental and nutritional point of view. These reasons make our study, and in particular this part of the experiment, relevant for addressing crucial features of policy regulation.

A contribution of this work is the inclusion in the analysis of children, since previous research is limited: we know from Dardanoni and Guerriero, 2021 that they are willing to pay for protection of environment, but little is known on their preferences on environmental and health characteristics. The other main contribution is providing evidences on similarities and difference between household members preferences.

This study opens the way for future research, and it is the pillar of a larger RCT (expected size: 400 families) involving broader age group of children. This work provided us prior for an efficient design of this larger study. We tested indeed the validity of the questions and the labels to be utilized.

Appendix

Consumer Survey (European Commission, 2011)

For studying the feasibility of the introduction of eco-labels on food, feed and drinks in the EU Ecolabel framework, they construct a survey with three objectives:

- to identify the influencing factors in the purchasing choices when consumers face environmental labels
- to analyse the risk of confusing the consumers among different labels
- to understand the power of EU Ecolabel in this sector compared with organic labels.

The survey was conducted in four countries, presenting heterogeneous stages in eco-labelling and labelling traditions.

From the report *Environmental Impact of Products* (EIPRO) it is possible to identify products with higher environmental impacts according to eight criteria:

- global warming potential
- human toxicity
- acidification
- ozone layer depletion potential
- eco-toxicity
- abiotic depletion
- photochemical oxidation
- eutrophication

However, essential issues are missing in the categories involved in EIPRO due to the difficulties in quantifying some environmental impacts. For instance, EIPRO does not consider fish stock depletion, wildlife protection, deforestation, water usage, landscape pollution.

Caveats of EC survey: done more than ten years ago, which can imply a significantly lower sensitivity compared to nowadays.

Personality traits

Parents PT	Question
I am usually the person who starts the conversations.	Extraversion
I see myself as a unique person who suggests new ideas.	Openness to Experience
I am a reserved person.	Extraversion
I see myself as an altruistic and helpful person.	Agreeableness
I know how to stay calm even in stressful situations.	Neuroticism
I tend to be a disorganized person.	Conscientiousness
I think that I usually worry a lot.	Neuroticism

from 1 to 5: 1 being don't agree at all, 5 being I strongly agree.

Children PT	Question
I usually play 1) on my own 5) with others	Extraversion
When I usually go to school 1) I'm worried 5) I'm calm	Neuroticism
I usually do my homework 1) willingly 5) unwillingly	Conscientiousness
When I'm able to help somebody 1) I help 5) I don't help	Agreeableness

from 1 to 5

TABLE 4.7: Personality Traits.

Mixed logit results

	(Collective) Choice	(Parents) Choice	(Kids) Choice
mean			
price	0.0157	-0.00491	0.0174
	(0.52)	(-0.23)	(0.75)
health	0.429***	0.360***	0.334***
	(6.11)	(6.30)	(6.70)
taste	0.505***	0.450***	0.561***
	(6.62)	(9.57)	(11.25)
environment	0.331***	0.242***	0.307***
	(4.24)	(4.95)	(5.84)
status_quo	1.564	0.839	2.538***
	(1.93)	(1.88)	(5.63)
standard_dev			
health	0.190	0.361***	0.0770
	(1.81)	(6.17)	(0.64)
taste	0.324***	0.241***	0.157*
	(4.00)	(4.32)	(2.48)
environment	0.325***	0.252***	0.183*
	(3.80)	(4.08)	(2.39)
status_quo	-3.009*	-1.740***	1.659***
	(-2.23)	(-4.63)	(4.94)
N	1512	2884	2352

t statistics in parentheses. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

TABLE 4.8: Mixed logit results with the prices non-random.

Mixed logit: mothers and fathers

	(Mothers) Choice	(Fathers) Choice
Mean		
health	0.327***	0.404***
	(4.84)	(4.63)
taste	0.560***	0.308***
	(8.54)	(4.80)
environment	0.141*	0.326***
	(2.31)	(4.40)
price	-0.0105	0.0171
	(-0.27)	(0.37)
status_quo	0.892	0.390
	(1.39)	(0.44)
SD		
health	0.240**	0.367***
	(2.81)	(3.95)
taste	0.235**	0.177
	(3.02)	(1.92)
environment	0.179	0.234*
	(1.79)	(2.04)
price	0.191***	0.186**
	(3.90)	(3.13)
status_quo	-1.797*	-2.641*
	(-2.29)	(-2.16)
N	1624	1260

t statistics in parentheses * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

TABLE 4.9: Mothers and fathers.

4.7.1 Mixed logit and interactions

	(1) Choice	(2) Choice	(3) Choice
health	0.401**	0.378***	0.351***
	(2.77)	(6.47)	(6.71)
taste	0.352**	0.459***	0.427***
	(2.97)	(9.15)	(9.05)
environment	0.295*	0.233***	0.147
	(2.36)	(4.72)	(0.78)
price	0.0269	0.00537	-0.00410
	(0.33)	(0.18)	(-0.15)
status_quo	0.811	-0.245	1.031*
	(1.72)	(-0.25)	(2.33)
env_IQ	-0.0144		
	(-0.56)		
h_IQ	-0.00585		
	(-0.20)		
taste_IQ	0.0243		
	(0.99)		
price_IQ	-0.00501		
	(-0.30)		
sq_educ		0.588	
		(1.27)	
env_envatt			0.0209
			(0.47)

t statistics in parentheses* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

TABLE 4.10: Mixed logit with interactions in the parents' sample.

	(1)	(2)	(3)	(4)	(5)
	Choice	Choice	Choice	Choice	Choice
health	0.427***	0.339***	0.410***	0.361***	0.361***
	(4.73)	(6.77)	(4.71)	(6.87)	(6.86)
taste	0.568***	0.567***	0.569***	0.590***	0.590***
	(6.33)	(10.90)	(10.93)	(11.02)	(11.04)
environment	0.198*	0.203*	0.192*	0.222	0.250
	(1.99)	(2.10)	(1.99)	(0.91)	(0.97)
price	-0.0153	0.0173	0.0185	0.0280	0.0279
	(-0.29)	(0.68)	(0.74)	(1.16)	(1.16)
status_quo	2.491***	2.487***	2.522***	2.566***	2.571***
	(5.25)	(5.39)	(5.46)	(5.33)	(5.35)
env_IQ	0.0530	0.0502	0.0568		
	(1.32)	(1.30)	(1.48)		
h_IQ	-0.0410		-0.0339		
	(-1.17)		(-1.03)		
taste_IQ	-0.000735				
	(-0.02)				
price_IQ	0.0158				
	(0.73)				
env_envatt				0.0197	0.0212
				(0.37)	(0.40)
env_class					-0.0109
					(-0.33)
N	2324	2324	2324	2232	2232

t statistics in parentheses * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

TABLE 4.11: Mixed logit with interactions in the children's sample.

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