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WP 2021.03

Suggested citation:

P. Quirion (2021). Output-Based Allocation and Output-Based Rebates: A Survey. *FAERE Working Paper, 2021.03.*

ISSN number: **2274-5556**

www.faere.fr

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Output-Based Allocation and Output-Based Rebates: A Survey

Abstract

Output-based refunding consists in distributing the value of taxes on pollution, or that of tradable emission allowances, to operators of emitting facilities, in proportion of their current production level. It is called output-based rebating in the case of taxes and output-based allocation in the case of tradable emission allowances. This practice is widespread, especially in climate policies, and has important economic consequences. We analyse these consequences, first in a deterministic setting and then accounting for uncertainty. While output-based refunding is detrimental to welfare in a deterministic, closed economy without prior distortions, it also provides some benefits. In particular, it is an efficient way to limit carbon leakage.

Then, we present the implementation of output-based allocation in the European Union, California, China, New-Zealand and Alberta, and discuss whether it should be maintained or phased out in the coming decades.

Résumé

Le remboursement basé sur la production (*output-based refunding*) consiste à distribuer la valeur des taxes sur la pollution, ou celle des quotas d'émission négociables, aux exploitants des installations émettrices, en proportion de leur niveau de production actuel. On parle d'*output-based rebating* dans le cas des taxes et d'*output-based allocation* dans le cas des quotas d'émission négociables. Cette pratique est très répandue, en particulier dans les politiques climatiques, et a des conséquences économiques importantes. Nous analysons ces conséquences, d'abord dans un cadre déterministe, puis en tenant compte de l'incertitude. Si le remboursement basé sur la production est préjudiciable au bien-être dans une économie déterministe et fermée sans distorsions préalables, il présente également certains avantages. En particulier, il s'agit d'un moyen efficace de limiter les fuites de carbone.

Ensuite, nous présentons la mise en œuvre de l'allocation basée sur la production dans l'Union européenne, en Californie, en Chine, en Nouvelle-Zélande et en Alberta, et nous examinons s'il convient de la maintenir ou de la supprimer progressivement au cours des prochaines décennies.

JEL codes: Q52, Q58
Keywords: output-based allocation; leakage; output-based rebate; output-based refunding; emission trading

Introduction

Many researchers have been defending for a long time the 'polluter-pays-principle' (OECD, 1975): the 'scarcity rent' created by environmental constraints should not be given to polluters but should be used to fund public goods or to reduce other taxes. For an emission trading system (ETS), this means that allowances should be sold (auctioned) rather than given for free to polluters; for a price instrument, that tax revenues should not be rebated to polluters, at least not systematically. Yet, many environmental policies implemented have failed to comply with that advice: early emission trading systems have overwhelmingly favoured free allocation (Boemare and Quirion, 2001), while revenues from early emission taxes have been often earmarked to fund abatement in polluting industries (Soares, 2012).

Various reasons explain this lack of application of the polluter-pays-principle, not least the political influence of polluting firms. In the context of greenhouse gas emissions, one of the main reasons put forward for free allocation of emission allowances has been the risk of carbon leakage. However, economic research has shown that not every type of free allocation is able to reduce this risk[1]; to have a direct effect on imports and exports, hence on leakage, free allocation has to be linked to current production, not to historical variables like past emissions (an allocation method generally labelled 'grandfathering') or past production (an allocation method generally labelled 'benchmarking', or 'static benchmarking'); cf. Quirion (2009); Sterner and Muller (2008).

Allocating emission allowances on the basis of current (and not past) production is generally labelled output-based allocation. Its equivalent for a price instrument is an emission tax whose revenues are rebated proportionally to current production (Sterner and Isaksson, 2006). This chapter intends to provide an accessible overview of the current state-of-the-art of academic research on output-based allocation of emission allowances, as well as output-based rebates. We will generally mention "Output-based refunding" (OBR) to encompass the latter two types of policies. The scope of the chapter includes allocation not fully proportional to output, e.g. with thresholds[2]

The rest of the chapter is organised as follows. We start by analysing the impact of OBR on economic welfare, abstracting from uncertainty (Section 1). In Section 2, we focus on the distributional impact of OBR. Uncertainty is introduced in Section 3, while Section 4 presents the (sometimes in impure form) implementations of OBR in some greenhouse gas emission trading systems. Section 5 concludes and discusses the perspectives for OBR, focussing on the European context.

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[1] Moreover, in assessments of climate policies based on general equilibrium models, a large part of leakage comes from the 'international fossil fuel price channel', which free allocation cannot address (cf. e.g. Fischer and Fox, 2012). In this chapter, we only deal with the other main leakage channel: the 'competitiveness channel', i.e. the fact that if unilateral climate policies increase production cost in trade-exposed emitting industries, they will lose market shares vis-à-vis foreign competitors, whose emissions will thus increase.

[2] For instance, in the third phase of the EU-ETS (2013-2020), operators of industrial installations received a given amount of allowances if their output was above 50% of the historical activity level (HAL), half of this amount if their output was between 25% and 50% of the HAL, one quarter of this amount if their output was between 10% and 25% of the HAL, and zero allowance if their output was below 10% of the HAL (Branger et al., 2015).

1. Welfare impact of output-based refunding in a deterministic setting: reducing leakage at the cost of output distortions

Abstracting from uncertainty and market power on the allowance market, an ETS is equivalent to a tax (Weitzman, 1974), so in this section we generically mention 'output-based refunding' (OBR) to encompass both output-based allocation (OBA) of emission allowances, and output-based rebating of tax revenues.

The first ex ante analyses of OBR and OBA, published in the late 1990s, do not always use this wording. Böhringer et al. (1998) label "grandfathering" an allocation method which is OBA as we define it; this is clear from their equation 2.1, where allocation is proportional to current, not past, production. Also, Parry and Williams (1999) model, in their own words, "performance standards", Edwards and Hutton (2001) "benchmarking and reallocation", Salmons (1999) "performance-based credit trading", all of which are also equivalent to OBR. To the best of my knowledge, Lashof (1997) and Fischer (1998, 2001) were the first authors to use in this context the expression "output-based allocation", which is now widespread.

These early papers already identify the key economic mechanisms at stake. Compared to a system in which allowances are auctioned or given for free based on past variables, OBR can be analysed as a subsidy to production, the value of which is the product of the allowance price (in monetary unit per tonne of CO_2-equivalent, thereafter CO_2e) by the benchmark (in tonnes of CO_2e per unit of output)[3]. Hence, for a given allowance price, OBR generates a higher production level than auctioning but does not change specific emissions (the emission level per unit of output), at least if these two decisions are separable[4].

Two consequences are noteworthy. First, with OBR compared to auctioning or a non-refunded tax, output is higher and specific emissions unaffected, hence aggregate emissions are higher, for a given carbon price. Hence, a higher carbon price is required to reach the same level of emissions. Second, since, absent other externalities (due e.g. to carbon leakage or market power on the product market), auctioning provides the first-best equilibrium by equalising the marginal abatement cost across abatement options, OBR is more costly than auctioning.

The easiest way to understand the latter result is perhaps to distinguish between process-level and product-level mitigation options. Think of the building material sectors: emissions may be reduced by process-level options (e.g. a more energy-efficient cement kiln) or by product-level options (e.g. replacing some cement by wood, if the latter is less CO_2-intensive). If allowances are allocated in proportion of current cement production, more cement is produced and the price of cement cheaper, compared to auctioning.

The problem is all the more acute that the allocation basis is thin – but such thin allocation basis is precisely needed to address leakage. Let us stick to the cement sector to illustrate this

[3] In other words, OBR acts like a combination of carbon pricing and output subsidy.

[4] They may not be separable; for instance, if the electricity generation fleet is composed of old plants with high specific emissions and variable costs, and of new plants with lower emissions and variable costs, then cutting electricity demand will drive some old plants out of the market, reducing specific emissions and changing the marginal abatement cost curve. However, we will stick to the separability assumption, like most of the literature.

point. More than 90 % of CO_2 emissions in the cement sector are due to the manufacturing of clinker, an intermediary product which has no other use, but which can be replaced by low-emission substitutes, to a certain extent. To quote Branger and Sato (2017), "*The incentive to use less clinker is dampened, however, if allowances are distributed in proportion to clinker production (clinker OBA). Conversely, if allowances are distributed in proportion to the output of the downstream product cement (cement OBA), producers can save emissions (and sell allowances) by importing clinker instead of producing it, thereby causing carbon leakage.*" This is what Demailly and Quirion (2006) called the "clinker dilemma". For cement, Branger and Sato (2017) propose a convincing hybrid solution: "*a clinker OBA modified with an allowances bonus-penalty depending on the clinker ratio (share of clinker in cement).*" However the same kind of trade-off exists for many products, for which such a solution is not granted. For instance, steel manufacturing must abandon the traditional production route based on coke, perhaps to move to hydrogen-based route (Vogl et al., 2018). With OBR it will be complicated to provide an incentive for hydrogen-based steel (which requires using the same benchmark for hydrogen-based steel as for coke-based steel) while preventing leakage, in the form of imports of steel scrap or pig iron (which requires *not* using the same benchmark for steel produced domestically with imported pig iron or steel scrap). In any case, it is hard to see how OBR could not favour process-level over product-level abatement options, e.g., lower-emission cement over wood.

However, OBR may provide a higher aggregate welfare than auctioning in two cases. Firstly, under market power on the output market, it may contribute to correct the inefficiently low output level (Gersbach and Requate, 2004). However, the effective subsidy provided by OBR is unlikely to be the optimal one (Fischer, 2011). Therefore, if public authorities aim at reducing the under-provision of output in some sectors, they should rather use less indirect policy tools.

Secondly, by increasing domestic output (compared to auctioning), OBR reduces foreign output, thus carbon leakage. Many simulations based on general (e.g., Böhringer et al., 2017) or partial equilibrium models (Demailly and Quirion, 2006; Monjon and Quirion, 2011) conclude that for a given carbon price, or for a given level of domestic abatement, OBR avoids a large part of the leakage which would arise under auctioning. This matters for CO_2-intensive, trade exposed industry such as steel or aluminium, but also for electricity, especially under sub-national policies like those implemented by some US states (Bushnell and Chen, 2012).

As a final note on carbon leakage, it may also happen vis-à-vis domestic, unregulated, emission sources, which also constitutes an argument in favour of OBR. Sterner and Isaksson (2006) analyse the Swedish charge on nitrogen oxides, which, like many environmental policies, applies only for installation bigger than a threshold. OBR reduces the incentive for polluters to switch to smaller emission sources, which would reduce the environmental effectiveness and raise the cost of the policy.

Whereas the benefits of OBR in terms of reduced leakage is higher or lower than the above-mentioned extra cost due to the imperfect balance between process-based and product-based abatement options remains an open question, though. The answer depends on the jurisdictions and sectors to which the climate policy would apply: OBR is more interesting if the products covered by the climate policy are highly substitutable with polluting, unregulated products, and less if they are highly substitutable with clean products. Since these elasticities of substitution are notoriously difficult to quantify, the answer depends on the functional forms and elasticities retained in the model used.

The first numerical assessment of this question conclude that the answer depends on the way international trade is modelled (Böhringer et al., 1998). Fowlie et al. (2016) find that OBR outperforms auctioning in terms of welfare for the US cement sector, part of the result stemming from the assumption of imperfect competition, generating an under-provision of output which OBR contributes to correct. On the opposite, Böhringer et al. (2007) find a negative impact of OBR on US welfare, compared to carbon pricing without rebating revenues, when applied to US energy-intensive trade-exposed industries. This being said, in their simulations, OBR performs better than a simple exemption of these industries, so it is still an option worth considering, especially if pricing carbon without rebating revenues is seen as politically difficult, if any for distributional reasons.

2. Distributional consequences of output-based refunding

Beyond the aggregate cost and welfare, OBR also differs from auctioning or other forms of free allocation by its distributional consequences, which may well matter more in terms of political feasibility. In this respect, results are more clear-cut: compared to auctioning or lump-sum (historical) free allocation, OBR boosts production and employment in the industries covered, and reduces the output price, which favours buyers of these industries. In manufacturing industries, those buyers are typically other industries rather than households, e.g. car manufacturers for steel. On the opposite, a large part of the electricity is consumed by households, and OBR is an efficient tool to limit electricity price increases for distributional reasons, as shown e.g. by Xenophon et al. (2019) for Australia and Burtraw et al. (2002) for the US.

Compared to lump-sum free allocation, OBA reduces profits in the industries covered, because it reduces the output price. Actually, lump-sum free allocation was implemented in most ETS worldwide precisely to maintain the profitability of the industries covered, so that these industries would not oppose the climate policies considered. However, this resulted in over-compensation, i.e. an increase in profits compared to what would have happened, absent the climate policies. In the EU ETS, the 'windfall profits' issue made the headline around 2006 (Sijm et al., 2006) and lead to the end of free allocation in the electricity sector, replaced by auctioning. Econometric evidence had shown that electricity suppliers passed the cost of the allowances on to their consumers (in non-regulated markets), even though they received them for free. Output-based allocation would have been another way to get rid of these windfall profits, but less efficient than auctioning, because for the European electricity sector as a whole, the limited interconnections drastically limit leakage, thus the rationale for OBA.

Compared to auctioning, the distributional impact of OBA obviously depends on the way auction revenues would be spent. Assuming that these revenues accrue the general budget, compared to auctioning, OBA favours profits in downstream industries, employment in the covered (and downstream) industries, with an ambiguous effect on profits in the regulated industries: on the one hand, the latter benefits from the free allowances but on the other hand, they suffer from a lower output price. Burtraw et al. (2002) in the case of the US electricity sector, find that "from the perspective of most electricity generators, the auction would be preferred to [OBA] as a method for allocating allowances". However, this is not a general result.

Finally, the allowance allocation method also impacts upstream industries. For instance, in Burtraw et al. (2002), a part of the negative impact of OBA on electricity generators' profit (compared to auctioning) stems from the fact that the entailed increase electricity production raises the natural gas price, which is an input of electricity generation.

To sum up, OBR should be favoured by stakeholders concerned by employment in the regulated industries (including, of course, trade unions), and by the upstream and downstream industries, but not necessarily by shareholders of the regulated industries, who benefit more from lump-sum allocation. Besides, compared to both auctioning and lump-sum allocation, OBR is likely to reduce plant closures, possibly a key point for policymakers.

3. Output-based allocation under uncertainty

The output level in many CO_2-intensive industries is difficult to predict, especially in the building material sectors (cement, steel, flat glass, bricks and tiles, etc.), which are very responsive to the business cycle. As shown by Branger and Quirion (2015), following the 2008-2009 crisis cement production in southern Europe has been approximately halved. Therefore, basing the amount of allowances allocated on current output impacts the situation of the covered industries.

However, it does not necessarily imply that under OBA the aggregate emission level varies with output, because public authorities may correct the allocation received by every installation so that the aggregate emission level is invariant. Following Meunier et al. (2017), I will label this alternative "Fixed Cap OBA", while with "Flexible Cap OBA", allocation is not adjusted for the aggregate output level, so aggregate emissions vary directly with output. Notice that in the EU ETS, while the amount of free allocation is linked to output (with thresholds, as mentioned above), the total cap is fixed[5], so the situation is closer to Fixed Cap OBA.

Under lump-sum allocation, a company covered by an ETS which suffers from an economic downturn benefits from excess allowances. Lump-sum allocation "smooths out" profit variations, while under both variants of OBA, this "insurance effect" does not exist (Quirion, 2009). However, if public authorities want to shelter industries from economic downturns, more direct and efficient ways exist, as those widely mobilised during the Covid-19 crisis.

Under Flexible Cap OBA, the aggregate emission level is uncertain, so this policy falls in the category of 'indexed regulations', which has been compared to price and quantity policies by several authors (e.g. Newell and Pizer, 2008; Jotzo and Pezzey, 2007; Quirion, 2005). In terms of global expected welfare, the uncertainty in aggregate emission level is rather inconsequential, since the marginal benefit curve for CO_2 abatement is almost flat. On the opposite, Flexible Cap OBA would reduce the variability of the allowance price hence of the

[5] In practice, the preliminary allocation is multiplied to a uniform 'cross-sectoral correction factor' (CSCF). In 2013 the CSCF was equal to 0.9427, then it declined by 1.74% per year.

marginal abatement cost. This may significantly reduce the expected cost of the policy, compared to a fixed cap.

The conclusions of the above-mentioned papers which address indexed regulations are not always in favour of these policies compared to a pure quantity one. The explanation is that they generally focus on nationwide policies, in which the emission level is indexed on GDP, itself only weakly correlated to baseline emissions. At the industry level, the correlation between output and baseline emissions is clearly higher and even very close to one for the main emitting sectors (for cement, cf. e.g. Branger and Quirion, 2015).

Meunier et al. (2018) compare Flexible Cap OBA to a pure quantity instrument (e.g., an ETS with lump-sum allocation), taking into account simultaneously the uncertainty about market fluctuations (e.g., output demand) and the fact that OBA distorts output away from the socially optimal level (cf. Section 1 above). In their model, a fraction of permits are allocated to firms based on their output while the rest is supplied in fixed quantity. They show that using a fraction of OBA is optimal despite the output distortions it creates. The intuition is that, although the increase in output is detrimental, it becomes of second order as the fraction of OBA permits goes to zero, while the benefits in terms of market stabilization is of first order.

Meunier et al. (2017) is a follow-up of the previous paper which includes leakage, and compares Fixed to Flexible Cap OBA. It concludes that a positive share of OBA is optimal, increasing with the sector leakage rate and with the market uncertainty. The latter result is particularly noteworthy since in existing ETSs, only proxies for leakage, not for uncertainty, are used to select the sectors in which OBA applies. Moreover, with linear marginal damage from emissions, Flexible Cap OBA dominates Fixed Cap OBA, because the former allows the permit price to stay closer to the marginal damage.

Admittedly, other, more direct solutions exist to stabilize the allowance price, especially a combination of price floor and price cap (Robert and Spence, 1976). Yet, some jurisdictions, notably the EU, have thus far refused to adopt a price floor, in spite of repeated calls from researchers (e.g. Flachsland et al., 2020). As shown by the ups-and-downs of the EU allowance price following the Covid-19 crisis, the recent Market Stability Reserve (MSR)[6] reform has not succeeded in stabilizing the allowance price. Moreover, introducing a price floor and a price cap (fixed at the ex-ante-optimal level) brings the ex-post equilibrium closer to the ex-post optimum but a gap remains. Meunier et al. (2018) show that even in this context, introducing a part of OBA increases the expected welfare.

To sum up, Meunier et al. (2017, 2018) conclude that from the expected welfare point of view, the price-stabilizing effect is a convincing argument in favour of Flexible Cap OBA. More precisely, it calls for using Flexible Cap OBA for a part of the allowances, combined with the distribution of the rest in fixed quantity.

[6] The MSR triggers adjustments to annual auction volumes when the total number of allowances in circulation is outside a predefined range: it adds allowances to the reserve by deducting them from future auction volumes if the total surplus is higher than 833 million allowances, and releases allowances from the reserve while adding them to future auction volumes ETS provided the total surplus is below 400 million allowances (European Commission, 2014).

However, for a jurisdiction which has announced an aggregate emission cap or budget, both Flexible Cap OBA and a price instrument generate a risk of exceeding its target, which may be politically problematic, at least in certain jurisdictions. These jurisdictions may then prefer Fixed Cap OBA, which keeps aggregate emissions constant. Regarding uncertainty Fixed Cap OBA behaves like lump-sum allocation or auctioning, except for the above-mentioned "insurance effect" related to idiosyncratic risks, which can hardly be seen as a relevant way to share risk among economic agents.

Finally, in the context of climate change mitigation, using a price instrument is likely to dominate both the quantity instrument and an instrument indexed to output (Flexible Cap OBA) (Quirion, 2005). Whether this price instrument is a tax with output-based refunding or another price instrument (tax, subsidy or combination thereof) does not affect the issues related to uncertainty which we have discussed here.

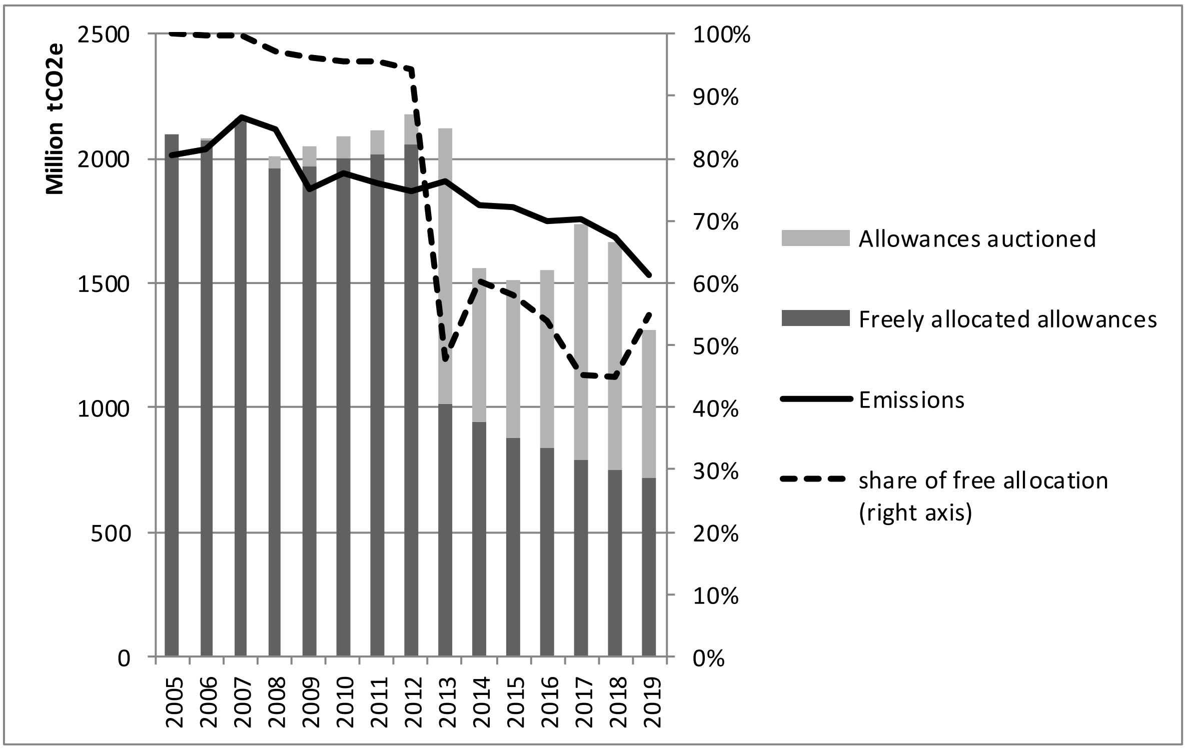
4. Output-based rebating in practice

a. A reluctant move toward output-based rebates: the EU ETS

Since its inception in 2005, the EU ETS has been the largest ETS in the world, by the amount and value of emissions covered. Until 2012, almost all allowances were distributed for free to historical CO_2 emitters, on the basis of rules which differed across Member States (Ellerman et al., 2010).

Since 2013 (phase 3 of the EU ETS), about half of the allowances are auctioned (Figure 1) and the rules for freely allocating the rest have been harmonised. As a general rule, electricity generation does not receive free allocation anymore, and for manufacturing industry, a distinction is made between sectors "deemed at risk of carbon leakage", which continue to receive free allowances, and the others, for which free allocation is gradually phased out.

Figure 1. Emissions, allowances freely allocated and auctioned in the EU ETS



Source: author's calculation, based on the European Environment Agency Data Viewer

For sectors at risk, allocation is the product of a benchmark (e.g. a number of allowances per tonne of clinker) multiplied by a historical activity level (averaged over five years). Thus, the basic rule does not include OBA. However, if current output drops below some threshold, allocation is reduced. These thresholds were introduced in order to reduce excess free allocation to low-activity installations.

In phase 3 of the EU ETS (2013-2020), these thresholds were relatively blunt: 10%, 25% and 50%. This means that an installation with an activity level at 51% of its historical level would have received 100% of its allowances, but only 50% with an activity level at 49%. For an activity level lower than 25% (respectively 10%), it would have received 25% (respectively 0%) of the allowances otherwise received (Branger et al., 2015).

Since 2021 (phase 4 of the EU ETS), the activity thresholds are smoother: the amount of allowances is adjusted proportionally if activity, averaged over two years, increases or falls by at least 15%, compared to the historical five-years average. Further adjustments take place if the difference exceeds the nearest 5% interval. Hence the EU ETS has progressively moved towards OBA.

Branger et al. (2015) have shown that in 2012, some EU clinker plants increased their production to stay above the thresholds, which boosted EU clinker production and emissions by around 5%. This behaviour happened in particular in Spain and Greece, which were severely hit by the 2008-2009 crisis. They find that the thresholds did reduce overallocation (by 6.4 million allowances) relative to a scenario without thresholds, but this gain was small compared to an output-based allocation method, which would have further reduced overallocation by 40 million allowances (29% of total cement sector free allocation).

The lower granularity of the phase 4 thresholds reduces the gain from this kind of gaming behaviour, since the change in allocation from crossing a threshold is either 5% or 15%. Yet, it also increases the number of installations which are close to a threshold, so the gaming behaviour has become more difficult to detect, but not necessarily less widespread.

b. New Zealand

The New Zealand ETS took effect on 1st January 2008. Unlike the EU and California ETS, it has no absolute emission cap and applies 'Flexible Cap OBA' to energy-intensive, trade exposed sectors, in order to prevent carbon leakage (Leining et al., 2020). According to Sense Partners (2018), "There is no evidence that the ETS has had impacts on New Zealand firms' competitiveness, in terms of impacts on profits or other measures of average commercial performance at the industry level." This is hardly surprising since allowances are output-based and the average allowance price has been lower than NZD10/tCO_2 (around €6/tCO_2) during the period analysed (up to 2016)[7]. A reform currently discussed will introduce, inter alia, auctioning under a fixed emission cap, a higher allowance price cap and a very gradual phase out of free allocation (New Zealand Ministry for the Environment, 2020).

c. Chinese pilots and national ETS

China announced its first-phase national ETS plan in December 2017, following several subnational pilot ETS. China's national ETS covers only electricity generation and applies 'Flexible cap OBA' type, as defined above. It employs multiple benchmarks for different types of power plants. Goulder et al. (2019) compare, in the case of the Chinese electricity sector, this design with a fixed cap and lump-sum allocation, and find that the cost of the former is about 47 percent higher. The reasons for this extra cost are that OBA causes producers to make less efficient use of output reduction as a way of cutting emissions (as explained above) and that the use of multiple benchmarks distorts the relative contributions of different power plants to emissions reductions. However, the authors also show that the use of OBA mitigates leakage and can reduce disparities in costs across technologies and regions of the country. Especially, it limits the costs in coal-producing regions. Since the ETS has not started operation before 2020, no ex post analysis has been performed.

d. California

In California, the ETS began in 2013 with coverage of electricity sold in the state (whether produced in-state or imported) and large-scale manufacturing. As in the EU, most allowances were initially freely allocated, but the share of auctioning increased progressively.

The California Air Resource Board allocates allowances to covered industrial facilities explicitly to minimize industrial emissions leakage[8]. The annual number of allowances allocated to an industrial facility is the product of four terms: an assistance factor, reflecting the leakage risk, a benchmark, a cap adjustment factor, decreasing by 4% per year, and the current output. Assistance factors were planned to be between 0 and 1 depending on the leakage risk in the sector, but were finally set at 1 for all sectors, presumably due to successful industry lobbying.

[7] Since then, the price has reached the cap currently fixed at NZD25/tCO_2.

[8] https://ww2.arb.ca.gov/our-work/programs/cap-and-trade-program/allowance-allocation/allowance-allocation-industrial.

CARB provides the final allocation for production in a given year in two parts: an initial allocation provided prior to the year, based on recent production data, and a true-up allocation provided after the year, which may be a positive or negative value, based on verified production.

To the best of my knowledge, no ex post assessment of the potential impact of the California ETS on carbon leakage from industrial facilities has been performed. It is worth noting that unlike in the EU, most of the discussion about carbon leakage in the California ETS deals with the electricity sector rather with the industry sector (Mastrandrea et al., 2020). This is due to the large electrical interconnections between California and neighbouring states, but also possibly to the output-based allocation method implemented for industrial facilities.

e. Alberta

In 2018, the Canadian province of Alberta introduced a new carbon pricing regime for large industrial emitters called the Carbon Competitiveness Incentives Regulation. All large industrial facilities (i.e. those with emissions exceeding 100,000 tonnes of carbon dioxide equivalent) are subject to an output-based carbon pricing system. To the best of my knowledge, there has been no ex post assessment of the impact of this policy so far.

5. Conclusion and way forward

While some environmental policies have been implemented before economic analysis took place, this is not the case with output-based refunding (OBR) applied to greenhouse gas emission control: the main economic insights had already been clearly stated in the early 2000s, before the implementation of the first such policies. OBR has clear drawbacks compared to lump-sum allocation or auctioning. In particular, it distorts against product-based abatement options, e.g. replacing cement by wood in construction. However, compared to lump-sum allocation or auctioning, OBR is efficient at preventing leakage, maintaining employment in regulated industries and limiting the distributional impacts. These pros explain the progressive adoption of OBR in the main ETS worldwide.

In the foreseeable future, carbon prices will continue to diverge, even across countries with similar incomes per capita. Besides, at least some word regions will hopefully trend toward deep decarbonation in manufacturing industry – the main application field of OBR. Against this background, the above-mentioned drawbacks of OBR will become more and more problematic. Long-term support of decarbonised processes in the manufacturing industry will remain necessary for a long time due to the persistent divergence in carbon prices across the globe. For instance, hydrogen-based steel is likely to remain costlier than coke-based steel in the long-run. However, this support will need to take other forms than free allocation (including OBR), since allowances need to be progressively phased out to reach climate neutrality. One form could be carbon contract for difference (CCfDs), which have gained prominent attention in recent years[9]. The stated aim of CCfDs is to compensate the extra cost

[9] A carbon contract for difference is a subsidy agreement between the regulator and a producer investing in a decarbonisation project. The volume of the annual subsidy payment is determined by the difference between the carbon price and a target price, times the abatement achieved by the proposed project (Vogl et al., 2021).

of low-emission processes, not to prevent leakage. Hence, in theory, they can be combined with OBR as well as with other leakage protection measures. Yet, CCfDs would exacerbate the main weakness of OBR: both favour process-level over product-level abatement options, e.g., lower-emission cement over wood.

There is thus a clear rationale for replacing OBR with auctioning combined to border adjustment, which does not generate the same distortions as OBR. Since this move may appear politically and diplomatically too difficult, an alternative is to keep OBR while complementing it with other policies, in order to incentivize the adoption of goods and processes with low carbon intensity. Indeed, it has been shown that complementing OBR with a tax on the consumption of carbon-intensive goods (whether imported or domestically produced) is equivalent to a combination of auctioning and border adjustment (Kaushal and Rosendahl, 2020; Neuhoff et al., 2016). Just as auctioning, this solution will face opposition from downstream industries (car manufacturing, for instance) because it would raise the price of carbon-intensive goods. Hence, an alternative policy would be to complement OBR with subsidies on less carbon-intensive goods (e.g. wood in construction, or other low-emissions binding materials in substitution to cement). This would be more costly overall but may generate less opposition. Overall, even though the drawbacks of OBR have been identified for twenty years, it may well continue to be used in carbon pricing policies in the coming decades.

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