European University Institute

ROBERT SCHUMAN CENTRE FOR ADVANCED STUDIES

FSR CLIMATE
Florence School of Regulation

EMISSIONS TRADING SYSTEMS WITH DIFFERENT LEVELS OF ENVIRONMENTAL AMBITION: IMPLICATIONS FOR LINKING

REPORT FOR THE CARBON MARKET POLICY DIALOGUE

LIFE DICET

RESEARCH REPORT

NOVEMBER 2020

AUTHORS
STEFANO F. VERDE, GIULIO GALDI, SIMONE BORGHESI,
JÜRG FÜSSLER, TED JAMIESON, EMILY WIMBERGER, LI ZHOU

QM-01-20-699-EN-N

ISBN:978-92-9084-936-0

doi:10.2870/801301

Cover photo by Arthur Ogleznev from Pexels

European University Institute

Badia Fiesolana

I – 50014 San Domenico di Fiesole (FI) Italy fsr.eui.eu

eui.eu

cadmus.eui.eu

Emissions trading systems with different levels of environmental ambition: implications for linking

Report for the Carbon Market Policy Dialogue

Authors: Stefano F. Verde, Giulio Galdi, Simone Borghesi, Jürg Füssler, Ted Jamieson, Emily Wimberger and Li Zhou

Abstract

This report was prepared to inform the Carbon Market Policy Dialogue (CMPD) between the European Commission, as the regulator of the EU Emissions Trading System, and the regulatory authorities for the emissions trading systems (ETSs) of California, Québec, China, New Zealand, and Switzerland. The report deals with the implications of linking emissions trading systems (ETSs) that differ from each other in the level of environmental ambition. The report provides a conceptual framework and summarizes the relevant scientific literature; it describes the current status of the six ETSs represented in the CMPD and, finally, it offers up a few ideas for discussion.

Keywords: Emissions trading systems; linking; environmental ambition; Carbon Market Policy Dialogue

Disclaimer:

The present report prepared by FSR Climate and its external collaborators is a deliverable of the LIFE DICET project, which is co-financed by the EU LIFE Programme of the European Commission. It reflects the authors' views and the European Commission is not responsible for any use that may be made of the information this report contains.

1 Introduction

Whenever the linking of two or more emissions trading systems (ETSs) is contemplated, their differences in environmental ambition are likely the first element that is considered. However, evaluating the convenience of whether and how to link ETSs that differ in environmental ambition is not a trivial task. For the jurisdictions involved, linking normally implies a change in the price of the emission allowances used in their own system. Depending on the magnitude of this change, the resulting price may or may not fall within a range considered ideal or acceptable by a jurisdiction. In addition, the same price change implies distributional effects within each jurisdiction, as well as financial transfers across jurisdictions: both things that could represent political difficulties. On the other hand, linking ETSs that differ in environmental ambition may make perfect sense. Linking systems that have different marginal compliance costs responds to the very same logic of an ETS: minimizing the cost of achieving an emissions reduction target by equalising marginal abatement costs. In fact, cost savings attained through a linkage increase with the difference in marginal compliance costs between linked systems. Moreover, the ambition of the systems taken together can be raised if the efficiency gain obtained from their linkage is leveraged for that purpose. On this last point, it would, then, be important to clarify from the outset what the goal of a linkage is. *Is it to increase the common environmental ambition of the linkers?* In the linking literature, the recurring metaphor of a person's choice of partner may fit here too, as the question is: *what do we want to achieve by being together?* Understanding this in the early stages of a relationship is generally desirable.

The purpose of this report – as that of the preceding introductory report (FSR Climate, 2020) and of the four others that will follow – is to inform the Carbon Market Policy Dialogue (CMPD).[1] In particular, the present report does four things: a) it provides a conceptual framework; b) it summarizes the scientific literature; c) it describes the current status of the six ETSs represented in the CMPD, namely those of California, China, EU, New Zealand, Quebec and Switzerland; and d) it offers up a few ideas for discussion. The hope is to stimulate the CMPD and, also, to provide relevant contents that will be taken up in the subsequent capacity building and dissemination activities within the DICET project (Deepening International Cooperation on Emissions Trading)[2]. Throughout, for the sake of simplicity, bilateral unrestricted linking is considered unless differently specified. The various forms of restricted linking are also relevant, but fall largely outside the scope of this report.

The report is structured as follows: Section 2 outlines the conceptual framework; Section 3 summarises the literature; Section 4 reports on the state-of-play and relevant experience of the six ETSs represented in the CMPD; and Section 5 offers a discussion and conclusions.

2 Conceptual framework

2.1 Defining environmental ambition

While a formal definition of the environmental ambition of an ETS does not exist, by ambition we generally mean the amount of abatement that an ETS promises to deliver. Accordingly, the ambition of an ETS may be assessed considering three dimensions: *emissions coverage*, *stringency* and *determinacy* – as we call it.[3]

By the 'emissions coverage' of an ETS we mean the share of a jurisdiction's total emissions that are regulated. Intuitively, an ETS that covers increasingly large shares of its jurisdiction's emissions indicates, all else being equal, an increasing level of environmental ambition. Similarly, an ETS can be considered more environmentally ambitious than other systems that cover smaller shares and are otherwise equivalent in the other relevant dimensions.

---

[1] https://lifedicetproject.eui.eu/carbon-market-policy-dialogue/

[2] https://lifedicetproject.eui.eu/

[3] For each of these dimensions, a relevant indicator may be considered and, if so, a mathematical formula is conceivable which would quantify environmental ambition by combining the three indicators in some way. Such a formula would reflect subjective preferences about the relative importance of the three dimensions.

The 'stringency' of an ETS refers to its targeted abatement level at a certain point in time or over a certain time period. It is expressed in percentage terms relative to expected business-as-usual (BAU) emissions, i.e. the emissions expected if the system was not in place. Estimates of BAU emissions and, therefore, of stringency, necessarily come with a margin of error. Thus, alternative metrics that are commonly considered are: a) targeted abatement relative to historical emissions; and b) allowance prices as a proxy for a system's marginal cost of compliance, i.e. the marginal cost of abatement for a given targeted abatement level. Between these two metrics, there are at least three reasons why allowance prices are preferable. First, allowance prices measure the actual economic pressure that an ETS alone exerts on regulated emissions. This is important because many exogenous factors determine regulated emissions, including other climate policies, economic growth and technological shocks. These factors affect allowance demand and, hence, allowance prices.[4] Second, the stringency of an ETS can vary over time as a result of changes in BAU emissions, and again allowance prices account for these variations. In this sense, allowance prices are a duly dynamic stringency indicator. Third, allowance prices allow direct comparisons of stringency between absolute-cap ETSs, a.k.a. cap-and-trade systems, and relative-cap ETSs, which impose a maximum carbon intensity relative to some measure of output (Ellerman and Sue Wing, 2003).

The last consideration leads to the third dimension of environmental ambition: what we call 'determinacy'. In this context, 'determinacy' is the quality of an abatement target to ensure emissions stay below a certain level irrespective of economic activity or, conversely, to accommodate lower or higher emissions depending on the economy's evolution. Some might argue that relative-cap ETSs are always, by definition, less environmentally ambitious than absolute-cap systems. The reason is that the former do not ensure that regulated emissions stay within predetermined limits if economic activity turns out to grow more than expected. However, while there is little doubt that the indeterminacy of the emissions outcome diminishes the environmental ambition of an ETS, it is debatable whether relative-cap ETSs are by definition less environmentally ambitious than absolute-cap systems. In principle, a relative-cap ETS whose stringency is higher than that of an absolute-cap system might be legitimately considered more ambitious (Sue Wing *et al*., 2008): it depends on the importance attributed to stringency and to determinacy. Indeed, a relative-cap ETS can be more stringent than an otherwise equivalent absolute-cap system (i.e. it can induce greater abatement) if economic growth in the former's jurisdiction is sufficiently strong and/or its constraint on emissions intensity is sufficiently tight.[5]

2.2 Environmental ambition and linking

Differences in environmental ambition between ETSs have economic, environmental and political implications for a potential linkage of the systems. The literature recalled in the next section analyses these implications. As a preliminary step, we discuss how the environmental ambition of an ETS, as previously conceptualised, relates to linking. Specifically, we clarify that not all the elements that are relevant for assessing environmental ambition are equally important in relation to linking.

When it comes to linking, differences in emissions coverage between ETSs are not relevant *per se*. Rather, differences in size matter, that is, differences in the absolute volume of regulated emissions. Size differences are a key determinant in the economic benefits that a jurisdiction can expect to attain by linking its ETS with another. In general, linking to a larger ETS, that is, one larger than other comparable systems, is economically convenient: as a net seller, a jurisdiction will access higher allowance prices and, as a net buyer, it will access lower prices (Doda and Taschini, 2017).

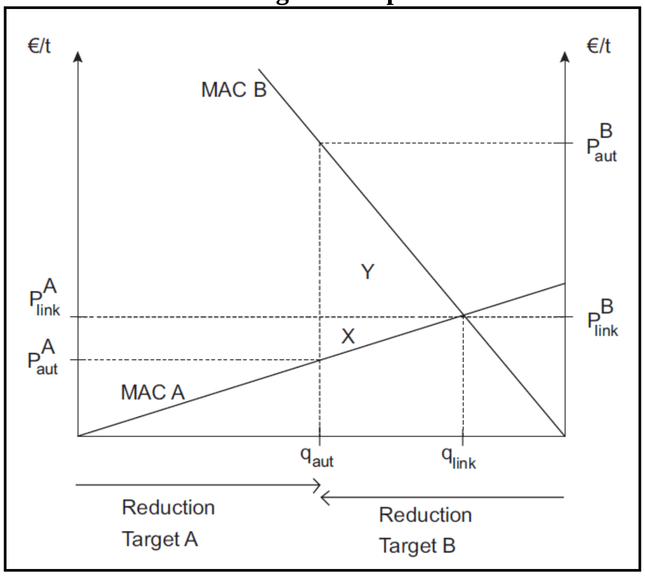
A second point is that differences in stringency between ETSs matter toward linking insofar as they translate into different marginal compliance costs. Differences in marginal compliance costs underlie the

[4] In a sense, targeted abatement relative to historical emissions is only a nominal metric, in that it does not account for the many factors other than the ETS itself which determine regulated emissions.

[5] Following Haites (2014), in an economy whose emissions are growing at 4%/year, an intensity reduction of 3%/year results in greater abatement (relative to BAU emissions) than a 2%/year absolute reduction does in an economy whose emissions are growing at 1%/year. Other things being equal, an intensity reduction of 6%/year results in even lower emissions.

main economic rationale for linking, which is to reduce the cost of total abatement, i.e. the sum of abatement produced by the systems. Differences in stringency between ETSs normally translate into differences in marginal compliance costs, and hence in allowance prices, but the relationship is not necessarily one-to-one given possible differences in abatement costs between jurisdictions.[6] In principle, it is possible to have equally stringent ETSs that result in different marginal compliance costs; and, conversely, ETSs that differ in stringency but that have similar marginal compliance costs. We illustrate this point with a graph borrowed from Flachsland *et al.* (2009).

Figure 1 – Differences in marginal compliance costs between ETSs.



In Figure 1, the two ETSs of jurisdictions A and B are expected to abate equal amounts of emissions (relative to BAU emissions). Assuming that they are equal in size, the two systems are equally stringent. Nevertheless, their pre-link marginal compliance costs and, thus, their (autarky) allowance prices differ ($P^B_{aut} > P^A_{aut}$). This is a case exemplifying why the relationship between differences in stringency and differences in marginal compliance costs is not necessarily one-to-one. That said, Flachsland *et al.* (2009) use the graph to show that the efficiency gain of linking originates from differences in marginal compliance costs. The area X+Y represents that kind of gain (more on this in the next section).

3 Literature review

The most relevant scientific literature for this report analyses the implications that differences in environmental ambition between ETSs have for their linking. It also covers the implications that linking itself, by potentially inducing strategic behaviour, has for the environmental ambition of a linked system. Implications of this kind, which regard the benefits, costs and risks of linking, both collectively and individually for the jurisdictions involved, are often classified as being economic, environmental and political in nature. The literature review is, then, structured accordingly.

3.1 Economic implications

[6] Differences in abatement costs between jurisdictions would reflect differences in the availability or cost of abatement technologies. Assuming that abatement costs vary by sector, differences in stringency will not translate one-to-one into differences in marginal compliance costs, and hence in allowance prices, also when ETSs differ in sectoral coverage.

3.1.1 The efficiency gain of linking

Differences in marginal compliance costs between ETSs provide the main economic rationale for their linking, namely, reducing the cost of total abatement.[7] As previously explained, differences in stringency normally translate into differences in marginal compliance costs, but this relationship is not necessarily one-to-one. Besides, differences in marginal compliance costs are deduced from those in allowance prices, though – it is worth recalling – equalisation of marginal compliance costs within an ETS rests on market efficiency assumptions about the allowance market.[8] With these caveats in mind, differences in allowance prices trigger trading between linked ETSs and cost savings are achieved as differences in marginal compliance costs between ETSs narrow. In standard partial equilibrium analysis, this is the net benefit that always comes with linking, making all jurisdictions better-off (regardless of distributional effects within jurisdictions).

We refer again to Figure 1 to illustrate the immediate mechanisms at play when two ETSs are linked together and how the resulting efficiency gain is distributed between the respective jurisdictions. When ETS A and ETS B are linked together, their pre-link allowance prices, $P_{aut}^{B} > P_{aut}^{A}$, converge to an intermediate level, which we call $\bar{P}$ ($P_{link}^{B} = P_{link}^{A}$). This level $\bar{P}$ is closer to the pre-link price in the system whose marginal abatement cost (MAC) curve is flatter (ETS A), whether because abatement is less expensive or simply because the system is larger.[9] Indeed, in the graph, just as differences in abatement costs, the relative slope of the MAC curves may reflect the relative size of the ETSs, the flatter curve corresponding to the larger system. Price convergence induces a shift in abatement efforts, from the jurisdiction where abatement is more expensive at the margin (B) to that where abatement is cheaper (A), $q_{aut} \rightarrow q_{link}$. The shift in abatement generates savings in total abatement costs which correspond to the area X+Y. Importantly, this efficiency gain increases with the initial difference in marginal compliance costs and with the size of the systems (Haites and Mullins, 2001). Moreover, its value partly accrues to B in the form of abatement cost savings (Y area) and partly goes to A in the form of revenue from sold allowances (X area).[10] The distribution of the efficiency gain depends on the relative slope of the MAC curves, with a greater share accruing to the jurisdiction (whether net seller or net buyer of allowances) for which the MAC curve is steeper.

3.1.2 Other economic effects

In partial equilibrium analysis, linking ETSs with different marginal compliance costs always generates an efficiency gain. This being the starting point, the first element to consider for a more realistic analysis is fixed costs. For example, the process of linking can require costly efforts, including negotiations over the alignment of technical requirements and of design features (Doda and Taschini, 2017).[11] If sufficiently large for a jurisdiction, these costs can discourage a bilateral linkage altogether or the participation of a jurisdiction in a multilateral linkage. A natural assumption is that a jurisdiction would only consent to a linkage if it can expect a net benefit from it.

Beyond administrative costs, a range of factors can diminish a jurisdiction's willingness or ability to link. A case in point are the distributional effects, between and within ETSs, that come with any linkage. As these are essentially political hurdles, however, we discuss them separately (Section 3.3). Likewise, possible concerns about the reduced environmental ambition that a linkage may cause are discussed in the next section. Other factors are economic in nature, but transcend the partial equilibrium framework

[7] Other fundamental economic rationales for linking: eliminating or reducing international competitiveness distortions related to differences in carbon prices, and creating more liquid and hence less volatile carbon markets.

[8] In an ETS, equalisation of marginal abatement costs through allowance trade only holds under market efficiency assumptions (Acworth et al., 2017; Hintermann *et al*., 2016; Flachsland *et al*., 2009).

[9] $\bar{P}$ will be equidistant if the systems are equal in size and also face equal abatement costs.

[10] Specifically, for B, Y is the difference between cost savings from lower abatement and the cost of emission allowances purchased from A. For A, X is the difference between the revenue from emission allowances sold to B and the cost of increased abatement.

[11] On the other hand, linking offers administrative benefits through mutual learning (Burtraw *et al*., 2013).

considered thus far. For a jurisdiction expecting to export allowances and hence to see allowance prices increase after linking, reduced competitiveness on international goods and services markets may be deemed more important than the revenue from exported allowances (Babiker *et al*., 2004; Copeland and Taylor, 2005). At the same time, for a jurisdiction expecting to import allowances after linking, the financial transfer associated with those, as well as reduced fiscal revenues from allowance auctions (insomuch as auctioning is used as the allocation method), may outweigh the expected benefit of reduced compliance costs. Besides, lower allowance prices may not help achieve sustainable development objectives in the domestic economy (Green *et al*., 2014; Green, 2017). These tradeoffs are qualitatively unaffected by smaller or larger differences in environmental ambition levels between ETSs that consider linking. However, the magnitude (in relative terms) of all effects increases, following a linkage, with the magnitude of the price adjustment.

Other economic benefits of linking include those promised by the enlargement of the allowance market, notably greater liquidity and reduced price volatility. As Doda and Taschini (2017) show, however, while price volatility can only decrease for two linked ETSs taken together (i.e. price volatility is equal or lower than on average under autarky), it might increase for one of them individually – if so, becoming an economic disadvantage for the corresponding jurisdiction. Price volatility after linking mainly depends on price correlation between ETSs. Moreover, the literature suggests that, for an absolute-cap ETS, linking to a relative-cap system entails greater volatility (compared to linking with an equivalent absolute-cap system). The reason is that in relative-cap systems allowances are partly distributed *ex-post*, thus causing liquidity spikes at the moment of adjustment (Sterk *et al*., 2006; Blynth and Bosi, 2004).

3.2 Environmental implications

The environmental implications of linking ETSs that differ in environmental ambition relate to the possible consequences for emissions. The question is whether differences in environmental ambition between ETSs that consider linking can lead to greater or smaller total abatement than if the same systems operated independently. The literature emphasises situations that result in lower total abatement, i.e. increased emissions. Notably, situations of this kind relate to linkages between absolute- and relative-cap ETSs and to strategic loosening of the stringency of an ETS. In any case, valid is the idea stressed by Mehling *et al*. (2018) whereby any economic gain that comes with a linkage also offers an opportunity for cooperatively increasing environmental ambition: *"Linkage is important, in part, because it can reduce the costs of achieving a given emissions-reduction objective. Lower costs, in turn, may contribute politically to embracing more ambitious objectives."*

3.2.1 Linking absolute- and relative-cap ETSs

Bilateral linkages between absolute- and relative-cap ETSs are somewhat problematic (DEHSt, 2013). The reason is that allowance trading triggers mechanisms whereby output in the jurisdiction with a relative-cap may increase and, as a result, overall emissions increase, too. Fischer (2003) shows that this is a likely outcome, regardless of whether the relative-cap system is net buyer or net seller. In the first case, output increases in the relative-cap system because abatement and thereby production costs fall. In the second, output increases because the output-subsidy effect of the increase in allowance prices outweighs the direct cost increase.[12] The same author, however, identifies a situation where this general result may not apply, namely if output from the two systems are substitutes or complements in the global market. Under such circumstances, cross-price effects may lead to reduced output in the relative-cap system, thus potentially eliminating or even reversing any increase in emissions. Addressing the same question (linking between absolute- and relative-cap ETSs), but using a different analytical framework, Marschinski (2008) finds that total emissions fall when the relative-cap ETS is a net seller. The difference with Fischer's (2003) general result is explained by the use of a different production function, namely one with increasing marginal costs, rather than a function with constant marginal costs. Furthermore, linkage leads to increased emissions if,

[12] Effectively, relative-cap ETSs, just as tradable performance standards, simultaneously impose a marginal cost to emissions and offer a subsidy to output (Fischer, 2001).

as a consequence of it, allowance prices in a net-seller relative-cap ETS reach a ceiling, so additional allowances are released (Haites, 2014).

On the whole, different outcomes are possible which depend on the specificities of the linkages under consideration. This suggests that numerical simulations with suitable models are necessary for evaluating specific linkages between absolute- and relative-cap systems.

3.2.2 Stringency as an outcome of linking

As far as total abatement is concerned, linkages between absolute-cap ETSs may seem unproblematic: abatement activity will partly shift across jurisdictions, but total abatement will be unaffected. However, the incentives that linking creates for adjusting a system's cap (or the efficiency target in a relative-cap system) may result in total abatement that is greater or smaller than under autarky. This has to do not with different design features of the ETSs, but with whether the respective jurisdictions cooperate to maximise total welfare. A number of studies analyse how the stringency of linked ETSs can be strategically adjusted by governments to their own benefit (Helm, 2003; Rehdanz and Tol, 2005; Carbone *et al*., 2009; Holtsmark and Sommervoll, 2012; Habla and Winkler, 2018; Lapan and Sidkar, 2019; Holtsmark and Midtømme, 2019). While results differ across studies, depending on model types and assumptions, they usually find that non-cooperative linking (in the sense specified) leads to less total abatement (i.e. higher emissions) than the same ETSs operated under autarky (Holtsmark and Weitzman, 2020). At a minimum, governments should preemptively agree on abatement targets when linking.[13] If such agreement exists, Flachsland *et al*. (2009) point out that the incentive to breach it (by altering stringency) can be weakened for several reasons. These can include: reputational damage; the threat of trade quotas or other penalties; as well as of the breakdown of cooperation in other policy areas.

3.3 Political implications

Differences in environmental ambition between ETSs that consider linking have important political implications. These are potentially decisive for a linkage to take place and its success. Political challenges arise with two types of distributional effects which accompany the efficiency gain of linking. One of these is revenue transfer between jurisdictions. For a jurisdiction that is net importer of allowances, substantial revenue transfers from the domestic economy into that of the exporting jurisdiction might not be politically acceptable.[14] The magnitude of these flows depend directly on the difference in stringency between the systems, as represented by their pre-link allowance prices (Burtraw *et al*., 2013). At the same time, changes in allowance prices after linking determine winners and losers within each jurisdiction: allowance buyers in the high-price ETS and sellers in the low-price system benefit from the link; conversely, allowance sellers in the high-price ETS and buyers in the low-price system suffer financial losses (Haites and Mullins, 2001). This kind of disparity of impacts may constitute a political barrier to linking depending on how strongly those who lose out lobby for their interests and depending, too, on how keen authorities are to overcome that opposition.

A second political factor that can weigh decisively against the realisation of a linkage, especially when the systems involved differ significantly in environmental ambition, is the partial loss of policy control over an ETS (Jaffe and Stavins, 2008). The limits to policy control regard both regulatory adjustments, which may be needed for a linked system to function properly and, above all, acceptance of co-determined allowance prices. Here we have prices that do not exclusively reflect domestic market conditions and which may, in some measure, deviate from levels considered preferable from a jurisdiction's own perspective. Substantially lower allowance prices post-link (i.e. compared to prices under autarky) may not, for example, be acceptable for a jurisdiction that greatly values carbon pricing as an approach to fostering low-carbon innovation (Flachsland *et al*., 2009). Importantly, the smaller the size of an ETS relative to the partnering system, the greater, in general, the loss of policy control for its authorities. On the

[13] As Green *et al*. (2014) put it, "linking without an agreement on targets would be like a monetary union between countries where each had the right to print money".

[14] Even if savings in abatement costs were larger in value, such transfers would be more salient.

other hand, the jurisdiction with a smaller system normally enjoys a greater portion of the efficiency gain from linking (Section 2.2).

The sum of the effects described leads to a paradox whereby the linkages that could yield the greatest benefits in terms of efficiency gains – by virtue of large differences in pre-link allowance prices – may also be politically the most difficult to implement (Ranson and Stavins, 2016; Zetterberg, 2012). Nevertheless, as Burtraw *et al*. (2017) emphasise, a large difference in allowance prices need not be an insurmountable barrier to linking. Various forms of restricted linking represent solutions that, while generally less advantageous in terms of efficiency gains, still provide long-term benefits.

4 Data from the ETSs in the Carbon Market Policy Dialogue

This section reports on the state-of-play and the relevant experience of the six ETSs represented in the CMPD.

4.1 California-Québec

4.1.1 State of play

California and Québec established independent Cap-and-Trade Programs in 2012, and formally linked Cap-and-Trade systems on January 1, 2014. Each jurisdiction maintains authority over their respective programs and work together to ensure the linked Cap-and-Trade Program achieves the greenhouse gas (GHG) emission targets of each jurisdiction through the transfer and exchange of fully fungible compliance instruments through joint auctions and trading.

Coverage of the California Québec Cap-and-Trade Program and emissions reduction targets

The California Québec Cap-and-Trade Program has a hard emissions cap that declines each year to achieve GHG targets established in each jurisdiction. The Program covers carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), sulfur hexafluoride (SF_6), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), nitrogen trifluoride (NF_3), and other fluorinated greenhouse gases. These covered gases are converted to carbon dioxide equivalent or CO2e. Currently, the California-Québec Cap-and-Trade Program covers approximately 85% of the GHG emissions of California and Québec. Each program has established emissions caps through 2030, with the cap declining each year by a factor outlined in the regulatory documents for each system. The Program begun with a limited scope in the first compliance period, 2013-2014. The limited scope covered the electricity and industrial sectors with a combined emissions cap of 186 MMT in 2013. The Québec cap represented approximately 10% of the 2013 Program cap. In 2015, the Program cap rose to 394.5 MMT CO2e as Program coverage expanded to include natural gas suppliers, transportation fuels, additional industrial sources. In 2015, the Québec cap represented 15% of the combined cap. From 2015-2020, the emissions cap declined by 3-4% each year. From 2021 to 2030 the combined cap declines by an average of 5% from 376 MMT in 2021 to 244.6 MMT in 2030. The cap in 2030 represents the emission reduction target separately established by each jurisdiction, 200 MMT in California and 44 MMT in Québec. Any change to in capped emissions across jurisdictions would require approval of both jurisdictions and subsequent approval of any subsequent program modifications. Emission caps beyond 2030 will be established based on the emission targets of each jurisdiction as they work towards carbon neutrality across their economies.

Allowance Price

The California Québec Cap-and-Trade Program has a firm aggregate cap on emissions and a fixed supply of emission allowances. Regulated entities must cover their reported GHG emissions with compliance instruments that include emission allowances and compliance offsets, which are verified GHG reductions that occur outside of capped sectors. Allowances can be purchased through joint quarterly auctions and through bilaterally trading by entities within and across jurisdictions. The Program is designed to deliver cost effective emission reductions to achieve the GHG targets for each participating jurisdiction.

The allowance price reflects the supply and demand for allowances. Given the uncertainty in allowance prices, the Program includes cost containment provisions to keep allowance prices within a range established through an auction reserve price and a price ceiling. The auction reserve price is established each year for the Program and increases 5% plus inflation annually. The 2020 auction reserve price is $16.68 USD or $22.03 CAD.[15] The price ceiling is maintained through the additional sale of allowances at pre-specified prices to covered entities in Québec and California for compliance.

While the California and Québec systems are harmonized across the majority of design features, there are a few areas in which jurisdiction specific mandates lead to different Program requirements for entities in California and Québec. In regard to the price ceiling, entities in Québec can purchase set aside allowances through sales of mutual agreement from the three tiers of the Allowance Price Containment Reserve. In California, entities can purchase allowances at reserve sales from two price containment points and a price ceiling.

In addition, while verified offsets are fully fungible across the Program, the treatment of offsets varies by jurisdiction. Entitles in Québec can cover up to 8% of their covered emissions through 2030 using offsets and Québec offsets are fully guaranteed. The offset limit for California entities varies from 8% in 2013-2020, 4% from 2021-2025, and 6% from 2026-2030. Offsets in California are also subject to buyer liability and from 2021-2030 50% of offsets must provide direct environmental benefits (DEBS) in the State of California. These variations do not affect the functioning of the linked Program. Other design features that impact the price of allowances and the cumulative cost of GHG reductions in the Program include allowance banking, multi-year compliance periods, and free allocation of allowances.

The allowance price in the California Québec Cap-and-Trade Program has tracked fairly closely to the auction reserve price over the course of 24 joint auctions. At the time of writing, the most recent joint auction was held on August 18, 2020 with 89% of all current vintage allowances sold at the auction reserve price of $16.68 USD or $22.03 CAD. Unsold allowances will remain in the auction account for 24 months at which time they can be purchased at subsequent joint auctions. Unsold allowances after the 24 month period will be retired. There has been discussion across the jurisdictions about the supply of allowances relative to demand and the resulting settlement price at joint auctions – the proceeds of which go fund additional GHG mitigation programs in each jurisdiction and provide benefits directly to consumers.

4.1.2 Relevant experience

Longevity

The California Québec Cap-and-Trade Program has been a successful bilateral, international linked ETS since 2014, with program roots going back to 2007. The Program has not thrived for the past decade by remaining static. The regulations that guide the California and Québec Cap-and-Trade systems have been modified to adapt to changing economic and political conditions. The Program has survived changing economic conditions, changing climate ambition, changes in governing administration and serves as an example of flexible, dynamic ETS. A lesson to be learned from the California Québec Cap-and-Trade Program is that a resilient ETS must be founded on strong uniform market principles with opportunities to revise existing Program details and be malleable in adapting to changing and unforeseen conditions.

Linking with other ETSs

The California Québec Cap-and-Trade Program has been a lesson in cooperation, partnership, and shared visions of linked climate ambition. The Program has its roots in the Western Climate Initiative (WCI) which was formed in 2007 when the Governors of the US states of Arizona, California, New Mexico, Oregon, and Washington signed an agreement to develop a regional target for reducing GHG emissions, track and manage GHG emissions in the region, and develop a market-based program to achieve the GHG target. By

[15] https://ww2.arb.ca.gov/sites/default/files/2020-08/aug_2020_summary_results_report.pdf

2008, two additional states and four Canadian provinces, British Columbia, Manitoba, Ontario, and Québec, joined WCI.

WCI built on existing GHG reduction efforts in the jurisdictions and in 2010, the 11 WCI jurisdictions released guidelines for developing a regional ETS. In 2011, WCI, Inc., a non-profit arm of WCI was formed to provide administrative and technical services to support ETS programs. The WCI guidelines and the services of WCI, Inc. form the basis of the California and Québec Cap-and-Trade systems. While participation in WCI has waned since 2007, the core principles of partnership and cooperation in addressing climate change and implementing joint strategies to reduce GHG emissions remains.

The California and Québec Cap-and-Trade systems were developed separately within each jurisdiction under the guidelines of WCI and utilizing the administrative and technical services of WCI, Inc. Since 2014, there have been other US states and Canadian provinces that have participated in WCI and WCI, Inc. Most notably, the California Québec Cap-and-Trade Program was bilaterally linked to the Canadian province of Ontario from January 2018 until the termination of the Ontario program later that year. The short bilateral linkage with Ontario provided many lessons in how shifts in administration can alter a jurisdiction's climate ambition as well as the mechanics of delinking programs.

The bilateral linkage of systems is guided by requirements outlined in the ETS regulation documents of each jurisdiction. For the California Québec Cap-and-Trade Program, linkage is part of a public regulatory process that requires amending the regulations of each jurisdiction as well as the signing of linkage agreements that commit the linked jurisdictions to work collaboratively to harmonize and implement a linked ETS as well as facilitate public release of appropriate information and confidentiality requirements for market sensitive data. While Ontario's swift departure did not result in noticeable market impacts, California and Québec subsequently bolstered WCI linkage requirements to ensure market stability in the event of future unanticipated delinking.

There have been other potential forays into linkage with the California Québec Cap-and-Trade Program. Nova Scotia is currently a member of WCI and while not bilaterally linked with California and Québec utilizes the services of WCI, Inc. There remains interest from other states and provinces in considering use of WCI, Inc. services and of linkage – both full bilateral and various degrees of linkage.

4.2 China

4.2.1 State of play

China has a goal of reducing its CO_2 intensity of GDP by 60-65% from the 2005 level by 2030, as its Intended Nationally Determined Contribution (INDC) under the Paris Climate Change Agreement. During the 12th and 13th Five-Year-Plan periods (2011-2015, 2016-2020), the national carbon intensity reduction targets *per* unit of GDP were set as 17% and 18%, compared to, respectively, 2010 and 2015. In the past ten years, China has made great achievements in green and low-carbon development. These achievements have benefited from the implementation of "carrot and stick" policies, in addition to the support of the Communist Party of China (CPC) Central Committee. The most important "carrots" comprise energy-efficiency subsidies for technology renovation and electricity price subsidies for renewable energy production. The most important "sticks" include, instead, the mandatory shutting down of outdated and excessive capacity in both the power and industrial sector, and the introduction of a series of energy efficiency standards for industrial production. The establishment of a national carbon market is a logical step in China's energy transformation and its development of policy tools to address climate change.

China's national ETS is, in the initial stage[16], a relative-cap ETS, with a flexible cap related to activity levels. It is actually a multi-industry tradable performance standard. The carbon emissions cap in China's carbon market is jointly determined by the performance standards for carbon emissions that reflect the carbon intensity reduction target and by economic output. The national emission trading market covers

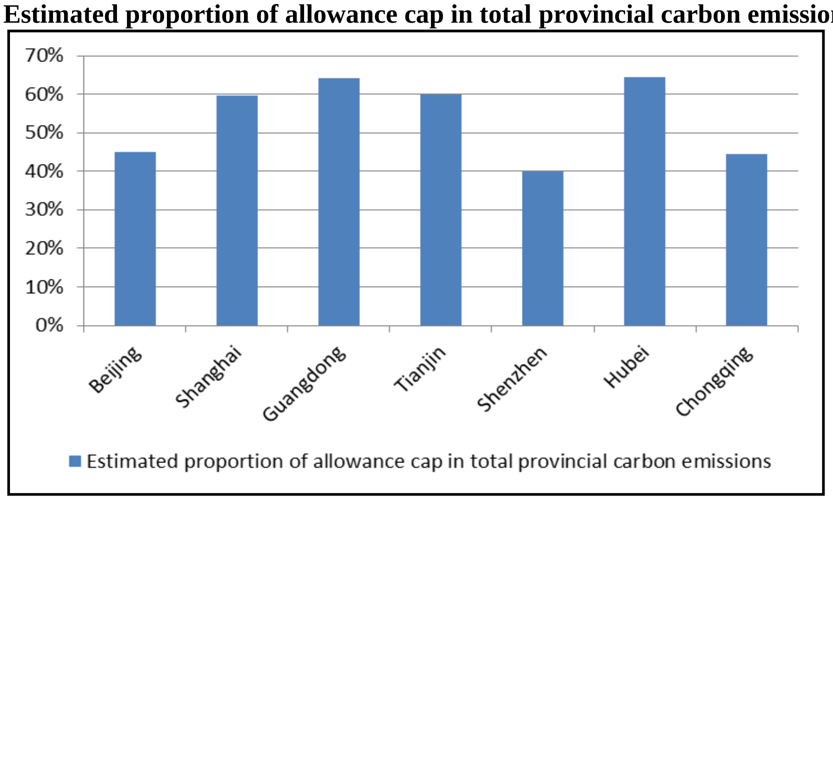
[16] The national ETS is supposed to become an absolute-cap ETS sometime in the future.

eight industries, which are main carbon emitting industries. It is estimated that about 7500 firms will be covered and national cap will be about 4.5 billion tonnes of CO_2, which accounts for some 70 percent of the country's total carbon dioxide emissions[17]. However, there is currently no detailed official allowance allocation method for all industries. Only for the power industry, there is the *Implementation Plan for the Allocation of CO_2 Emission Allowances for Key Emission Entities in the Power Industry (Including Captive Power Plants and Cogeneration Plants) (Draft) in 2019*[18] with suggested benchmark values for trial allocation. At the time of writing, the final plan is still under discussion and comments and suggestions are being collected internally.

As these are early days for ETS in China, all system and design elements will need to be improved. At this early stage, linking may further increase the uncertainty of China's ETS for future development. It is hoped, though, that there will be a linking between the China's ETS and other ETSs in the future. China's ETS can either be fully connected to ETS in other countries or regions, or they can be connected in a restricted or partial way. China can also start with a one-way connection, establish cooperation with other systems, and gradually move to double-way connections, thus providing a stable transition period for connections between systems.

As of 2018, Beijing, Tianjin, Chongqing and other provinces had set a clear target for reaching peak emissions as part of their implementation plans or programs for controlling GHG emissions during the 13th Five-Year-Plan period. Beijing, for example proposed to reach peak emissions by 2020 or sooner and this target seems likely to be met; for Tianjin, the target is around 2025. Meanwhile, during the 12th and 13th Five-Year-Plan periods, China has broken down the national carbon intensity reduction target *per* unit of GDP to the provincial level. For the seven pilots, economic growth and uncertainties have been fully taken into consideration, and cap setting has been combined with provincial carbon intensity reduction target.

Figure 2 – Estimated proportion of allowance cap in total provincial carbon emissions in 2017.



[17] Initially, only the power sector is going to be regulated. After that, the ETS will be extended.
[18] China's Ministry of Environmental Protection (CMEP) (2019). A series of training on Carbon market allowance allocation and management, 2019.

Figure 2 shows estimated proportion of allowance cap in total provincial carbon emissions in 2017, based on published allowance cap by provincial government[19,20,21,22] and total provincial carbon emissions estimated from the energy balance sheet[23]. It can be seen that the pilot carbon markets choose a relatively large coverage and were expected to achieve certain carbon reduction.

The carbon market has made a positive contribution to carbon emission reduction in the pilot areas. Carbon emissions from enterprises in the scheme decreased 6.05% in 2015 and 2.59% in 2016 compared with the levels in 2014 in Wuhan. In 2013 and 2014, the total carbon emissions of regulated enterprises in Beijing decreased by, respectively, 4.5% and 5.96% *per* year, with a cumulative emission reduction of 6.3 million tonnes. Over 2016-2018, the carbon intensity of covered enterprises in the Beijing carbon market has been cumulatively reduced by 16.5%. Compared with 2011, the total carbon emissions of enterprises in the scheme in Shanghai decreased by 11.7% in 2014. Compared with 2010, the absolute amount of carbon emissions from enterprises in the scheme in Shenzhen decreased by 12.6%, and carbon emission intensity decreased by 34.2% in 2014 compared with 2010[24].

4.2.2 Relevant experience

Building a harmonized national carbon market covering multiple regions and industries and letting the market play a role in driving emission reductions will effectively reduce the financial pressure on the government and achieve NDC targets at the lowest societal economic costs. According to the calculations from the Institute of Energy Environment and Economy of Tsinghua[25], China's commitments under the *Paris Agreement* can be reached through the development of a national carbon emission trading system and will save about 0.1% of GDP in 2020 and 0.6% in 2030 compared to the scenario without ETS.

As the national ETS is under construction and as the total cap related with relative benchmark values and actual production will not be available until compliance has finished, the ambition level of national ETS is still uncertain and as such is difficult to evaluate. The benchmark values are key parameters for ensuring the rationality of the cap with adequate shortage. There are also some benchmark discussions in the Study on Coverage, Cap Setting, Allowance Allocation Methodologies and Supplementary Mechanisms project [26], which is supported by the World Bank's China Partnership for Market Readiness. For the power sector, the verification data of historical carbon emissions from 2013-2015 submitted to the National Development and Reform Commission (NDRC) by more than 1,700 power generation enterprises (including combined heat and power – CHP) in 36 provinces and cities are under research. There are 11 types of turbines with different fuels, scale and technologies. It is impossible to set a single benchmark value for all units in China in this, the first stage.

---

[19] China Environment News. Hubei province: to promote the construction of carbon emission trading and carbon market (in Chinese) [EB/OL]. 2019.2.13 http://epaper.cenews.com.cn/html/2019-02/13/content_80330.htm

[20] NetEase News. When will China's carbon market unify. Beijing officially launch carbon emission trading in 2017(in Chinese) [EB/OL]. 2016-11-22. http://www.tanpaifang.com/tanjiaoyi/2016/1122/57700.html

[21] Fujian provincial department of ecology and environment. Announcement by Fujian provincial department of ecology and environment on the carbon emissions and allowance compliance of key enterprises in 2018(in Chinese) [EB/OL]. 2019-07-18. http://www.tanpaifang.com/zhengcefagui/2019/071864747.html

[22] Chongqing development and reform commission. Chongqing development and reform commission issued a notice on carbon emission allowances for 2017(in Chinese) [EB/OL].2018-3-9 https://tpf.cqggzy.com/news/notice/147.html

[23] National bureau of statistics of China. China energy statistics yearbook 2018 [M]. Beijing: China Statistics Press (in Chinese).

[24] China Environment News. Shenzhen enterprises participate in carbon trading enthusiasm to improve the overall level of energy consumption (in Chinese) [EB/OL]. 2016.12.01. http://www.tanpaifang.com/tanjiaoyi/2016/1201/57782.html

[25] Institute of Energy, Environment and Economy, Tsinghua University. Study on air quality and Health Impacts of the National Carbon Emission Trading Market (internal report) [R]. Beijing.2018.

[26] Institute of Energy, Environment and Economy, Tsinghua University. World Bank PMR Project - China Carbon Market "Research on Coverage, Cap Setting, Allowance Allocation Methods and Supplementary Mechanism" (in Chinese) [R]. Beijing. 2018.

Approach 1: 11 benchmarks. All thermal power units were divided into 11 categories according to their technical parameters and each has its own benchmark value.

Approach 2: three benchmarks. All thermal power units were divided into conventional coal-fired units, unconventional coal-fired units and gas-fired units, and each category of units has its benchmark value.

Approach 3: 4 benchmarks. All thermal power units were divided into: conventional coal-fired units above 300MW; conventional coal-fired units with 300MW or lower; and unconventional coal-fired units and gas-fired units. Each category of units has its benchmark value.

The calculation formula of allowance for a unit is as follows:

$$A = A_e + A_h$$

In which

A—The total allowance of the unit, tCO_2;

A_e—The allowance for electricity supply, tCO_2;

A_h—The allowance for heating supply, tCO_2;

For each kind of unit, the allowance is calculated based on benchmarks for electricity supply of the unit, benchmarks for the heat supply of the unit and correction coefficients for cooling mode and heating ratio.

The allowance for electricity supply is calculated as follows:

$$A_e = Q_e \times B_e \times F_l \times F_r$$

In which

Q_e—Power supply of the unit, MWh;

B_e—Benchmark for the electricity supply of the unit, tCO_2/MWh;

F_l—Correction coefficient for cooling mode;

F_r—Correction coefficient for heating supply.

The allowance for heating supply is calculated as follows:

$$A_h = Q_h \times B_h$$

In which

Q_h—Heat supply of the unit, GJ;

B_h—Benchmark for the heating supply of the unit, tCO_2/GJ.

In current discussions, the third approach has been deemed to be most appropriate and about a 1% shortage is expected when setting benchmark values.

The pilot carbon markets in China have now been running for some time and as such offer some lessons. Based on pilot data from 2013 to 2015, the total allowance slack of the pilot ETSs could be usefully considered in judging emission reduction effectiveness[27]. Comparing the emissions reduction rate between 2005-2010 and 2010-2015, the pilot ETSs in Guangdong, Beijing, Tianjin, Hubei and Shenzhen can be seen to have reduced additional emissions by the implementation of ETSs, and can drive local industry to take more actions in reducing emissions. In the Chongqing carbon trading pilot, its allowance allocation scheme has played a role in driving enterprises to reduce carbon emissions. But the scheme does not, in its own right, actually produce 'additional' emissions reduction. Based on the provincial panel data, from 2005 to 2016, the emissions reduction effects of pilot projects were studied using the Difference-in-Differences (DID) model. The results show that: (1) the implementation of a carbon trading policy significantly reduces the total emission (24.2%) of industrial CO_2 in all seven carbon emission trading pilots[28]; (2) the implementation of carbon markets in Beijing, Shanghai and Hubei had a significant inhibitory effect on

[27] Wang *et al*. (2018).
[28] Zhang *et al*. (2020).

local carbon emissions, but a promoting effect in Guangdong, and no significant effect was found in Tianjin; (3) In terms of time-lag effect, no significant inhibiting effect on local carbon emissions was found during the first year of the operation of the Shanghai carbon market. The effect, however, grew significantly from 2015 to 2016; while for Hubei and Beijing, a significant effect was shown from the pilot's first year[29].

The impact on industrial sub-sectors has been evaluated. The causal impact of China's pilot ETSs on reducing carbon emissions at the initial stage (2013–2015) is explored, with a DID model[30]. It is revealed that China's pilot ETSs had a statistically significant negative impact on carbon emissions, the carbon emissions of the ETS-covered sub-sectors and total carbon intensity. And this impact has presented an overall enhanced trend according to year-by-year analysis by applying PSM-DID estimation. The impact on carbon emissions grew from 2013 to 2015.

Taking industry as an example, DID, PSM-DID and SFA were used to investigate the impact of ETSs on industrial carbon emissions and carbon intensity, based on data from pilots and non-pilot provinces and cities, from 2008 to 2014[31]. First, it was found that ETSs has significant inhibitory effects on industrial carbon emissions and carbon intensity, reducing respectively 4.8% and 5.2%. ETS also increased energy technical efficiency and energy allocative efficiency and its influencing mechanism was related to energy technical efficiency. When the emission-abatement effect from energy technical efficiency proved greater than the emission increase effect from energy allocative efficiency, the ETS policy eventually achieved emission reduction.

4.3 European Union

4.3.1 State of play

Coverage of the EU ETS and emissions reduction targets

The EU ETS is a classical cap-and-trade system: it imposes an absolute cap on regulated emissions, which decreases annually by a predetermined linear reduction factor. The system regulates carbon dioxide (CO_2), nitrous oxide (N_2O) and perfluorocarbon (PFC) emissions from about 11,000 large installations, which include power stations and heavy energy-using industrial plants (oil refineries, steel works and production of iron, aluminium, metals, cement, lime, glass, ceramics, pulp, paper, cardboard, acids and bulk organic chemicals), as well as from domestic flights.[32] In total, the EU ETS covers about 45% of EU's GHG emissions. Over Phase IV (2021-2030), the cap will follow a steeper trajectory than in the previous trading periods, reaching in 2030 a level that is 43% below that of emissions in 2005. However, in the context of the EU Green Deal – what promises to be a historical re-launch of EU climate policy – and the connected climate neutrality target, whereby net-zero emissions are to be achieved by 2050, amendments of the EU ETS are under consideration to further increase its environmental ambition (European Commission, 2019, 2020a). Achieving climate neutrality by 2050 would require increasing the EU's GHG emissions reduction target for 2030 to at least 55% below 1990 levels, up from the current 40% (European Commission, 2020b). For the EU ETS, this implies a further tightening of the cap. The extension of the EU ETS perimeter to previously uncovered sectors – namely shipping, road transport and the building sector – is also being considered.

Allowance prices

---

[29] Yi *et al*. (2020).
[30] Zhang *et al*. (2019).
[31] Guangming and Zhang (2017).
[32] Both to limit administrative costs and to avoid disproportionately burdening small firms, in most sectors only installations above certain production capacity thresholds are subject to the EU ETS. As regards aviation, only flights within the European Economic Area are currently subject to the EU ETS.

Allowance prices are a key proxy for the stringency of an ETS (Section 2.1). After a long downward trend, which set off in 2008, prices of EU allowances (EUAs) quickly recovered in the last three years. Mainly due to the impact of the 2008-2009 economic crisis on industrial output, and the absence (at that time) of a mechanism to adjust allowance supply, the EU ETS allowance market saw the accumulation of a massive surplus. Excess supply reached over 2 billion of allowances in 2013, which roughly corresponded to one year's volume of regulated emissions. Neither the withholding of 900 million allowances (a.k.a. 'backloading') nor the establishment of the Market Stability Reserve (MSR), a new mechanism for correcting market imbalances, had strong effects on allowance prices. This is presumably because both interventions were originally intended to adjust supply only temporarily. Conversely, in late 2017, political agreement over the reform for Phase IV (2021-2030), which included a tightening of the cap and a strengthening of the MSR, had an immediate, durable impact on allowance prices *via* expectations (Quemin, 2020). Starting from €4/tCO_2, allowance prices approached €25 in less than a year. They then fluctuated without dramatic variations, until the Covid-19 pandemic struck the European economy in March 2021. The market has been highly volatile since then, a result of uncertainties about the economy's recovery and prospective adjustments to the EU ETS as part of the EU's Green Deal. At the time of writing (mid-July 2020), EUA prices have broken €30 for the first time since April 2006.

4.3.2 Relevant experience

The Market Stability Reserve

One of the main lessons from the European experience with emissions trading is that an ETS, especially one with an absolute cap, should be equipped with a mechanism for possible supply adjustments, if its stringency is to be preserved over time. If the economy grows differently from what was expected when the cap was set, or if any relevant unanticipated event occurs, baseline emissions deviate from their originally expected levels. This has obvious consequences for allowance demand and allowance prices, as well as for the level of abatement produced by the system and its cost over the long term. As explained above, the EU ETS ran into just this type of problem, and the establishment of the MSR was a structural response to it. Several studies analyse the functioning of the MSR (e.g., Hepburn *et al*., 2016[33]; Perino and Willner, 2017; Flachsland *et al*., 2020), which only started operating in 2019, but none to our knowledge does it specifically in relation to linking. Yet, the question deserves consideration because the MSR has distinct features from most if not all price-control mechanisms currently used in other ETSs (Osorio *et al*., 2020).

Linking with other ETSs

Since its inception in 2005, the EU ETS has been a building block of the international carbon market and a catalyst for its expansion. While this function has been mainly carried out through linkage with the Kyoto Protocol's Flexible Mechanisms (Clean Development Mechanism and Joint Implementation), there have also been attempts to directly link the EU ETS with other similar systems. Some of these attempts were successful, others failed.

In 2008, Norway, Iceland and Liechtenstein joined the EU ETS via a multilateral linkage. The three countries joined the EU ETS by virtue of being partners in the European Economic Area. Norway had already its own national ETS, which had been operating since 2005 (Klemetsen *et al*., 2020). Full integration required limited regulatory adjustments by the Norwegian ETS (its design was similar to that of the EU ETS) and, on the EU side, modest temporary concessions in the application of EU ETS legislation (Ellerman *et al*., 2010). In 2009, the year of the UNFCCC COP15 (Copenhagen), the EU led an ambitious project which aimed at creating an OECD-wide carbon market. The plan was eventually abandoned, however, after the proposal for a nationwide ETS was rejected by the US Senate. In 2010, negotiations started between the EU and Switzerland for linking their ETSs. The agreement on a bilateral linkage was

[33] This is the editorial introducing a special issue of the *Journal of Environmental Economics and Management* dedicated entirely to the economics of the MSR.

signed in 2017 and the linkage itself started fully operating in January 2020. The bilateral linkage between the EU ETS and the Swiss ETS was the first of its kind for the EU and, more generally, it was the first for two Parties under the Paris Agreement. Finally, in 2012, the European Commission and the Australian government announced their intention to link their ETSs. The first stage of linking was supposed to occur in 2015 with a one-way (unilateral) link, whereby covered entities in Australia would have been able to use EUAs to fulfill up to 50% of their compliance obligations. The linkage would have become bilateral three years later. However, following Australia's national elections, in 2013, the linking plan fell apart, as the new Australian government decided to repeal ETS legislation altogether.

4.4 New Zealand

4.4.1 State of play

The ETS from its start until 2012

New Zealand legislated to establish an ETS in 2008, after a two-year process of policy debate and public consultation. The design of the proposed ETS was set out in a detailed consultation document published in September 2007. The ideas behind the decision to adopt emissions trading, and the design features of the ETS, are set out in that document. These ideas and the already-established views that drove them can be traced back to the negotiation and adoption of the Kyoto Protocol. There was an established view that all of the six Kyoto gases could be equated and traded off against one another, using a single metric of CO_2-equivalent based on hundred-year global warming potentials. A second established view at that time was that forestry emissions and removals could be regarded as equivalent to fossil fuel emissions and that these could also be traded off against each other.

On this basis, a single measure would be used in the New Zealand ETS and internationally through the Kyoto Protocol flexibility mechanisms. This in turn would allow complete fungibility and a liquid market, broader than just the small market that could develop domestically in New Zealand, and the most economically efficient approach to reducing emissions and to increasing removals. The government's overall approach to climate policy emphasised the fundamental importance of pricing emissions, and envisaged a less important role for complementary measures aimed at specific sectors and market failures.

For most Annex 1 countries, ideas and assumptions about the equivalence of all emissions and removals are of secondary importance. Most of their emissions are fossil CO_2. Their forests are relatively stable and do not contribute a large proportion of a national inventory, either in terms of emissions or removals. These issues are more important for New Zealand because:

- CH_4 and N_2O from livestock farming (mostly CH_4) make up half of New Zealand's CO_2-equivalent emissions. The role of CH_4 as a biological emission source, and as a potent but short-lived greenhouse gas in the atmosphere, is a matter of ongoing scientific and political interest.

- Emissions and removals from plantation forestry also make up a substantial part of New Zealand's inventory, with removals offsetting up to a third of the country's emissions in some years. Significant net emissions may occur at other times.

The twin ideas that all emissions and removals were equivalent, and that a broad tradeable price measure was the essential and economically efficient basis for reducing emissions, determined many features of the ETS. The ETS was seen as ultimately providing a single, uniform price that would apply to all emissions and removals in every sector. Recognising practical and political constraints, the government set out proposals that would extend coverage over five years, starting with forestry and energy emissions and covering all significant emissions from 2013. All emissions except agricultural CH_4 and N_2O were brought into the ETS between 2008 and the end of 2012.

To reduce compliance costs, and consistent with the intent of complete coverage, the points of obligation for energy emissions were placed upstream. Energy suppliers surrender emission units equal to the potential emissions from use of the fuel that they sell. All coal, gas, electricity, and liquid fuel users in New Zealand have ETS costs incorporated in their energy bills, regardless of their size or what sector of the economy they operate in.

Deforestation in the five years before 2008 meant that forestry emissions and removals were seen as an urgent issue. The ETS placed a cost on deforestation, and incentivised plantation forestry, from 2008. It applied the rules of the Kyoto Protocol for forestry emissions and removals, by accounting separately for established (pre-1990) forests and new forests established after that time. Forest owners who established and maintained new forest would need to bank units to cover emissions from harvesting, but they would realise a net surplus of units for sale if their removals exceeded their emissions over time.

The ETS market was expected to bring together forest owners, with a surplus of removal units to sell, and the fossil energy and industry sectors who bought them to meet surrender obligations. The availability of CERs and ERUs for import and surrender provided a reserve supply, while some surplus forestry units could also be exported and sold for voluntary or compliance use in other Annex 1 countries. This model worked as anticipated until 2012 and the fall in Kyoto unit prices.

The Paris Agreement and new approaches to policy

Between 2013 and 2015 emission prices were low both internationally and for New Zealand ETS participants, and only minor changes were made to the ETS; from 2016 prices rose until they were effectively limited by the fixed price option (price ceiling) at NZ$25.[34]

Like other countries, New Zealand put forward its first Nationally Determined Contribution (NDC) in 2016. The NDC was set after a long process of policy development, economic modelling, and public consultation.

A new government came to power in November 2017 with a commitment to introduce 'Zero Carbon' legislation which would set targets for New Zealand to reduce its domestic emissions in line with the temperature goals of the Paris Agreement, and set up the legal and institutional framework for the transition that would be needed.

The Zero Carbon Act was passed in November 2019, with broad support across nearly all parties in Parliament. The Act set a national target for 2050, with five-year national emission budgets and plans to manage the transition needed to reach the target. The Act also established the independent Climate Change Commission to advise the government on the budgets and reduction plans, and on ETS settings. The legislated target takes a 'split gases' approach. It requires emissions of all greenhouse gases other than biogenic CH_4 to reach net zero by 2050. Biogenic CH_4 from livestock farming and waste, which makes up 41% of New Zealand's current CO_2-equivalent emissions, will not be required to reach zero; there is an interim target of a 10% reduction by 2030.

The government followed this up with legislation, presented to Parliament in October 2019, to reform and update the ETS. This Emissions Trading Reform Act was passed into law in June 2020. It represents a comprehensive reform of the ETS, including a process for setting caps on the supply of units in line with national budgets to be set under the Zero Carbon legislation.

The Act sets up a process for ETS settings – including the supply of units, price control measures, and potentially allowing and regulating international trading – to be set and revised over time. To cover the interim period before this can happen, the government has also proposed provisional settings which will give the market guidance on the likely unit supply and price controls up to 2025. The government has also published estimates of mitigation costs, which will inform price expectations.[35]

[34] Equivalent to US$22.23 or €14.31 at time of writing.

[35] https://www.mfe.govt.nz/publications/climate-change/marginal-abatement-cost-curves-analysis-new-zealand-potential-greenhouse

The Act will also simplify the accounting approach for plantation forestry. Outside the ETS, there is now a significant new policy in the form of the 'One Billion Trees Programme' which is supporting new forest planting through direct grants and partnerships with landowners and communities.

4.4.2 Relevant experience

Outcomes for the ETS in its original form

The ETS achieved one of its aims immediately on introduction by changing the economics of plantation forestry. Ultimately this meant that New Zealand met its Kyoto Protocol target for 2012, and is likely to meet its target for 2020, with a substantial contribution from the maintenance of existing plantation forestry and establishment of new forest.

The New Zealand ETS was the first to put reporting and surrender obligations on forestry and on some other sectors including waste disposal and refrigerants. This was achieved with the use of a well-resourced and effective registry, carefully designed methods, simple and usable on-line reporting tools, and a staged approach that allowed methods to be carefully developed for each sector.[36] The result was a high level of compliance and relatively few problems in implementation.

The link to the Kyoto system, and an unquantified right to access eligible Kyoto units for compliance, were set in legislation and could only be changed by the Parliament. The government wanted to prioritise economic recovery in the years following the global financial crisis, and low emission prices were seen as a necessity for allowing businesses to recover. In combination, these factors meant that the ETS continued to allow the surrender of low-cost Kyoto units until early 2015.

The intended outcome was to keep the cost to business low in the short term. An unintended consequence was to increase the number of units that many ETS participants were able to bank for future surrenders. Over 100 million Kyoto units were surrendered for the 2012-14 compliance years, which allowed participants to bank a similar number of New Zealand Units (NZUs). NZUs are not vintaged and can be surrendered at any time.

The ETS in a new policy environment

Many of the technical features of the ETS are still fit for purpose, are unaffected by new legislation, and are likely to be retained for the future. These include the registry systems, the upstream points of obligation, the option of voluntary participation, which has been used by large energy users, e.g. the iron and steel sector, and the emission reporting methods.

The Emissions Trading Reform Act is intended to update the ETS and make it an effective tool for managing and reducing domestic emissions in line with domestic targets. Putting caps on the supply of units year by year, and ensuring that the amount distributed through auctioning and free allocation for industrial entities does not exceed the cap, is basic to achieving this. Every year the government will be required to set and announce caps five years ahead. The Bill sets out a process that has significant flexibility to adjust the announced caps. When they are set and announced in a particular year (say this is year *Y*):

- The caps for years $Y + 1$ and $Y + 2$ are fixed and can only be adjusted under very limited circumstances including use of the price control measures or a *force majeure* event
- The caps for years $Y + 3$ and $Y + 4$ can be adjusted to ensure proper operation of the ETS
- A new cap will be set and announced for year $Y + 5$.

This flexibility may avoid problems like the 'waterbed effect' that could result when economic change or the effect of other policies make pre-set caps obsolete, resulting in price instability and the frequent use of price control measures. If the legislation is applied as anticipated, the clear decision-making processes and qualitative restraints in the law will ensure that any use of its flexibility does not compromise environmental integrity.

[36] https://www.eur.govt.nz/Authentication/Logon.aspx?ReturnUrl=%2f

Proposed settings 2021-25

The government released a consultation document in December 2019 (consultation closed at the end of February) which proposed a provisional national emission budget and ETS settings for the five years 2021-25.[37] On 2 June 2020 the government announced the budget and ETS settings that will be put into effect by regulation.

There is significant uncertainty about New Zealand's projected net emissions.[38] Estimates range from 355 Mt to 369 Mt, depending on the economic impact of Covid-19. The government has set a provisional budget for the five years of 354 Mt of CO_2-equivalent. This will require abatement of 15 Mt in relation to the most likely business-as-usual scenario.

The total cap on ETS supply for these five years is 159.5 million NZUs, equal to the budget less the projected 194 Mt emissions that are not covered by the ETS. However the supply of non-forestry units will be only 132.5 million, because units are being withheld to address the issue of over-supply.

The ETS is currently over-supplied with units that have been banked from previous periods. A total of 132 million NZUs are in circulation, and about 54 million of these units are held by entities that will not require them for future surrender obligations. The government is withholding 27 million NZUs, which will force a reduction of about half the current over-supply over five years.

The units supplied will comprise:

- 42.9 million free industrial allocation
- 89.6 million sold at auctions.

Auctions will be held every three months, with the first auction planned for 17 March 2021. The quantity given as free industrial allocation, which is indexed to production, can vary from year to year. The number of units auctioned will be adjusted each year to account for this.

These settings have been based on an analysis of abatement costs and projections of emissions and removals. However, they will remain provisional. The Climate Change Commission will be given the task of carrying out more extensive analysis and may recommend revised settings in future. The numbers will also need to change if emissions of CH_4 and N_2O from livestock farming enter the ETS by 2025.

Other issues for the future of the ETS

The government has made an agreement with the agriculture sector to work together to develop new policies and measures to manage and reduce agricultural emissions. The Emissions Trading Reform Act legislates for the future inclusion of agricultural CH_4 and N_2O in the ETS, but only as a back-up option. If it proves feasible to develop and implement alternative price measures that meet government requirements, the ETS may not be extended to include these emissions.

Any future consideration of CH_4 will also be complicated by the differential treatment of CH_4 in New Zealand's long-term domestic emissions target. It may no longer be appropriate for CH_4 and other gases to be treated as being fully fungible in the ETS.

Changes in international and domestic policy for forestry will also affect the ETS over time. Under the new legislation, the ETS rules for plantation forestry will change to use 'averaging' as the method to account for emissions and removals. This will mean that forest owners will not have to carry out all of the monitoring that is needed to measure carbon stocks and account for carbon over time. Reporting will be simpler than in the past. Forest owners will receive units only for establishing a new forest, and surrender units only if they deforest their land or change management practices. These changes will make the ETS a more effective incentive for new forest planting.

[37] https://www.mfe.govt.nz/consultations/nzets-proposed-settings

[38] Emissions less any forestry removals.

However, the incentive provided by the ETS sits alongside other issues for land use. Other incentives, including the One Billion Trees Programme, and regulations are important for ensuring that forests are located on suitable land and that their social and environmental effects are managed.

4.5 Switzerland

4.5.1 State of play

The Swiss CO2 Act

The first CO2 Act entered into force in 2000 and was replaced by a revised version in 2013. This is the current foundation of Swiss climate policy. It provides that, by 2020, at least 20 percent of GHG emissions as compared with 1990 levels must be reduced through domestic measures.

The policy instruments to implement the CO2 Act are:

- A CO2 levy (currently at CHF 96/tCO2 – about EUR 90/tCO2)
- Energy efficiency target agreements with industry (in particular for installations too small for the ETS)
- Emission trading system
- Climate cent levy and CO2 emissions compensation of transport-related fuels

The CO2 Act is currently under revision in view of the 2021-2030 phase. A parliamentary decision is due in autumn 2020, and it is expected that the CO2 Act will have to pass a popular referendum in spring 2021.

The Swiss ETS

The first commitment period of the Swiss ETS lasted from 2008 to 2012. Companies were incentivized to participate voluntarily in the ETS by being exempted from the CO2 levy as a result. Companies received freely allocated allowances according to agreed emissions reduction targets, which were negotiated on the basis of technological potential of economic viability measures to reduce GHG emissions within the company. Around 450 companies participated voluntarily in the scheme and accepted emissions targets. The commitment was considered fulfilled if they surrendered the quantity of emissions allowances required to offset their effective CO2 emissions by 1 June 2013.

For the period 2013-2020, the Swiss ETS[39] was completely redesigned. It is now a cap-and-trade system that is to a large extent a copy of the ETS of the European Union (EU ETS). Therefore, Swiss companies are subject to the same rules as their competitors in the EU. Based on historical activity data, an absolute quantity of emission allowances is determined in the system (‘cap’). For each ETS participant, the same benchmark values as in the EU ETS are used to calculate the quantity of emission allowances that are allocated free of charge based on the installations activity level during a base period. The rules are virtually the same as in the EU ETS.

Emission allowances are freely tradable (‘trade’) and can be surrendered to the Confederation to cover the greenhouse gases emitted or sold to other ETS participants. For 2013, the cap was 5.63 million tonnes CO2eq and has decreased annually by the same absolute amount (1.74% of the 2010 baseline) to around 4.9 million tonnes CO2eq in 2020. Companies participating in the Swiss ETS have to report their annual GHG emissions to the Confederation and surrender the necessary emissions allowances to cover them.

[39] See FOEN 2019: Swiss climate policy, online: https://www.bafu.admin.ch/bafu/en/home/topics/climate/info-specialists/climate-policy.html [14 July 2020].

The Swiss ETS currently includes GHG intensive companies from the cement, chemical and pharmaceutical, refinery, paper, district heating, and steel sectors (among others). As of 2020, slightly over 50 installations are included in the Swiss ETS.

There are about CHF 2-3 million of revenues *per* year from ETS auctions by the government. These revenues are added to the general government budget. Before the linking with the EU ETS in early 2020, the secondary market was not very liquid, which is one of the reasons that several stakeholders pushed for linking the Swiss ETS to the EU ETS.

4.5.2 Relevant experience

In 2017, the Swiss Federal Audit Office, an independent governmental entity which regularly assesses the efficiency of the policies and instruments of the Swiss government, carried out an evaluation[40] of the climate change mitigation impact of the Swiss ETS. Its comprehensive analysis of the Swiss ETS and its interaction with other exemption possibilities such as the target agreements[41] was to identify its impact and possible efficiency losses (empirical *ex-post* analysis). The study combined literature reviews, interviews with stakeholders, quantitative analysis and the work with case studies. The report found that the Swiss ETS had generated little incentive to reduce emissions in the businesses in the scheme, because of excess free allocation of emission allowances to most entities.

5 Discussion and conclusions

A necessary premise for almost any type of comparison of the six ETSs represented in the CMPD, including a comparison of their environmental ambition levels, is that these systems are today in different evolutionary stages. On the one hand, the ETSs of California, Quebec, EU and Switzerland are mature systems, as they have been operating for several years now, and have fundamental qualitative characteristics that are stable and similar to each other. On the other hand, there are the ETSs of New Zealand and China, which are currently in phases of structural (re)definition. A process of profound reform of New Zealand's ETS has almost come to an end. The most important effect of this reform is that the NZ ETS will also become, from 2021, a cap-and-trade system, like the other four mentioned above (the system has never had its own cap).[42] As regards the Chinese ETS, not only does it have yet to start operating and some of its fundamental parameters have yet to be defined (first and foremost, the benchmarks for the allocation of emission allowances), but it will be, at least for a few years, a relative-cap system. This in itself does not constitute an insurmountable obstacle for potential linkages with ETSs that have an absolute cap, but since this type of linkage can result in higher emissions (i.e. total emissions could increase as a consequence of the linkage, compared to the levels they would reach if the systems remained independent)[43], the scenario – that of a full linkage, at least – seems unlikely.[44,45]

An element that further complicates our comparison of the six systems is the fact that the environmental ambition of an ETS is itself not a univocally defined concept and that different metrics are commonly used to quantify it. We have then suggested that environmental ambition can be thought of and

---

[40] SFAO 2017: Evaluation der Lenkungswirkung des Emissionshandelssystems, online: https://www.efk.admin.ch/images/stories/efk_dokumente/publikationen/evaluationen/Evaluationen%20(51)/16393BE.pdf [15 July 2020]

[41] Small installations that cannot join the Swiss ETS may enter into target agreements i.e. with the help of an agency for emissions mitigation identify economic migitation measures and define their individual emission reduction target that is verified by an independent third-party expert. Installations that adhere to their target agreement are then excempt from paying the CO2-levy.

[42] Still today, the NZ ETS does not have its own cap.

[43] See Section 3.2.1.

[44] In the case of the EU ETS, a linkage with a relative-cap system is ruled out even by law (art. 25 of the EU ETS Directive).

[45] For linkages between absolute and relative-cap ETSs, forms of restricted linking are more plausible (see, e.g., Li et *al.*, 2019).

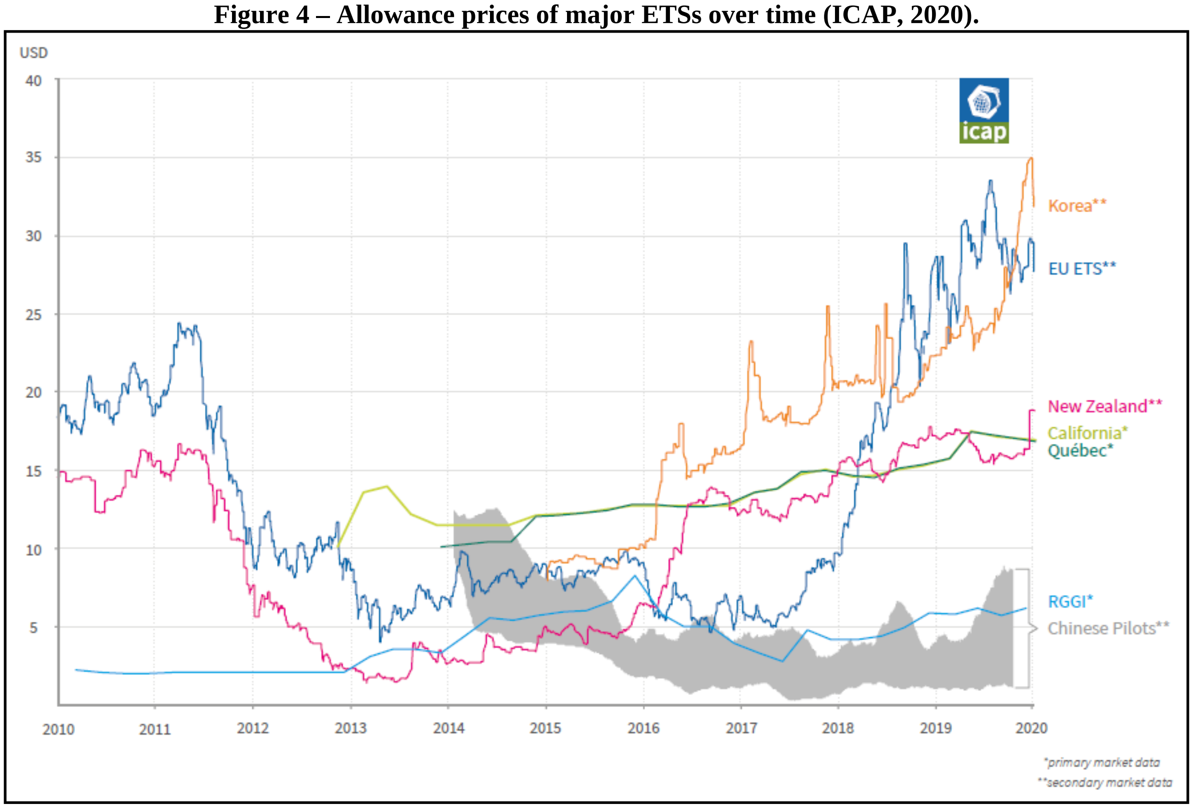
assessed considering three dimensions of a system, namely emissions coverage, stringency, and determinacy (see Section 2). In the context of linking, the stringency dimension assumes particular importance. In principle, the stringency of an ETS determines the marginal cost of compliance and, hence, the price of allowances. In turn, differences between ETSs in marginal compliance costs (together with the size of the systems) determine the efficiency gain of a linkage. Therefore, insofar as they do reflect differences in marginal compliance costs, differences in allowance prices between ETSs are a key indicator of this possible benefit. The paradox, highlighted by the literature, is that the greater the efficiency-gain potential the more difficult it is to accomplish a linkage. The distributional effects between and within jurisdictions that accompany a linkage represent one of the main obstacles in this sense.

In Section 4 of the report we have provided basic information about the state-of-play of the six ETSs in the CMPD. In the same vein, the graphs in Figure 3, taken and adapted from ICAP (2020), report some relevant data. The diamond graphs show, for the ETSs of the EU, Switzerland, California and Quebec, recent values of four key parameters, three of which are directly relevant to environmental ambition (two more specifically to stringency): the share of the jurisdiction's emissions that falls under the system (*coverage*); the average yearly decline rate in the system's cap between 2017 and 2020 (*cap trajectory*); and the average allowance price in year 2019 (*allowance price*). Since allowance prices refer to 2019, that is, before the linkage between the EU ETS and the Swiss ETS was finally agreed, price values are not the same for the two systems. As prices will converge, once the two systems are technically connected, the EU ETS and the Swiss ETS can be regarded as a pair of linked ETSs comparable to that of California and Quebec.[46] In general, there are some numerical differences between the systems (between and within pairs). But, again, the main economic benefit of a linkage between ETSs emerges precisely by virtue of the existence of initial differences in marginal compliance costs and, therefore, in allowance prices.

[46] To operationalise the link, the emissions trading registries of Switzerland and the EU need to be linked electronically. Due to the COVID-19 crisis, the start of operation of the electronic link, initially planned for May 2020, was postponed to September 2020. Initially, the registry link will perform transfers between the Swiss and EU ETSs only at set times. For more details, see: https://www.bafu.admin.ch/bafu/en/home/topics/climate/info-specialists/climate-policy/emissions-trading/the-swiss-emissions-trading-registry--ehr-.html

Figure 3 – Key metrics of four well-established systems (ICAP, 2020).

Figure 4 – Allowance prices of major ETSs over time (ICAP, 2020).



Of course, in any ETS, allowance prices vary over time for different reasons (Figure 4). Nevertheless, the EU ETS and the Swiss ETS, on the one hand, and the ETSs of California and Quebec, on the other, seem today – after years of operation and effective reform – as stable and well aligned pairs with respect to the timing of decarbonisation.[47] Though the aspects to consider when evaluating the opportunity to link ETSs are always many, the elements observed today (i.e. systems not identical but on the whole comparable, different prices but not excessively so, and similar long-term emissions reduction targets) represent important preconditions. It would therefore seem an appropriate moment to start evaluating the possibility of linking the four systems, by simulating economic impacts, by analysing the legal implications, etc.

Moving to an intertemporal dynamic perspective,, part of the literature on linking (reviewed in Section 3) focuses on the effects that a linkage can induce in terms of strategic behaviour. This literature shows the importance of agreeing, when a linkage is negotiated, on the future emission reduction targets of the systems involved. This would clearly serve to exclude the possibility of subsequent unilateral changes in the stringency of an ETS, which could have unwanted repercussions on connected systems.[48] We add that agreements of this type would allow interested parties to verify directly whether a linkage between their ETSs could be a lever to raise environmental ambition considering the savings it would bring. Given the well-known gap between the climate mitigation objective of the Paris Agreement and the sum of the NDCs in terms of expected mitigation, the potential for increasing environmental ambition is a compelling rationale for linking ETSs (Mehling *et al*., 2018).

Finally, we note that while it is certainly appropriate to agree on future emission reduction targets when a linkage is being negotiated it would also be desirable to agree on a fluctuation band for allowance prices. Indeed, the absence of an agreement on this aspect would leave it open to a jurisdiction to manipulate allowance prices to its advantage, notably by making the system more lenient, so as to export more allowances. Without altering its emission reduction targets, a jurisdiction could do so by modifying the policy mix that affects the emissions regulated by its ETS. Harmonizing the mechanisms for controlling allowance prices and agreeing on their fluctuation limits is, therefore, not a separate issue from that of agreeing future mitigation targets.

[47] California is committed to achieving climate neutrality by 2045. The European Union aims to achieve the same goal by 2050.

[48] Still, unilateral deviations from an agreement may occur. So, including de-linking provisions in a linking agreement would be important too (see, e.g. Pizer and Yates, 2015, and Borghesi and Zhu, 2020).

References

Acworth, W, Ackva, J., Burtraw, D., Edenhofer, O., Fuss, S., Flachsland, C., Haug, C., Koch, N., Kornek, U., Knopf, B. and M. Montes de Oca (2017), *Emissions Trading and the Role of a Long Run Carbon Price Signal: Achieving cost effective emission reductions under an Emissions* Trading System, International Carbon Action Partnerhsip, Berlin.

Babiker, M., Reilly, J. and L. Viguier (2004), *Is international emissions trading always beneficial?*, The Energy Journal, 25(2), 33-56.

Blyth, W., and M. Bosi (2004), *Linking non-EU domestic emissions trading schemes with the EU Emissions Trading Scheme*, Organisation for Economic Co-operation and Development (OECD) and International Energy Agency (IEA), Paris.

Borghesi, S. and T. Zhu (2020), *Getting married (and divorced): A critical review of the literature on (de)linking Emissions Trading Schemes*, Strategic Behavior and the Environment, forthcoming.

Burtraw, D., Munnings, C., Palmer, K. and M. Woerman (2017), *Linking carbon markets with different initial conditions*, RFF Discussion Paper 17-16, Resources for the Future, Washington, D.C.

Burtraw, D., Palmer, K., Munnings, C., Weber, P. and M. Woerman (2013), *Linking by degrees: incremental alignment of cap-and-trade markets*, RFF Discussion Paper 13-04, Resources for the Future, Washington, D.C.

Carbone, J. C., Helm, C. and T.F. Rutherford (2009), *The case for international emission trade in the absence of cooperative climate policy*, Journal of Environmental Economics and Management, 58(3), 266-280.

Copeland, B.R. and M.S. Taylor (2005), *Free trade and global warming: a trade theory view of the Kyoto protocol*, Journal of Environmental Economics and Management, 49(2), 205-234.

DEHSt (2013), *Linking different emissions trading systems – Current state and future perspectives*, Deutsche Emissionshandelsstelle, Umwelt Bundesamt.

Doda, B., Quemin, S. and L. Taschini (2019), *Linking permit markets multilaterally*, Journal of Environmental Economics and Management, 98, 102259.

Doda, B. and L. Taschini (2017), *Carbon dating: when is it beneficial to link ETSs?*, Journal of the Association of Environmental and Resource Economists, 4(3), 701-730.

Ellerman, D., Convery, F., and C. de Perthuis, C. (2010), *Pricing carbon – The European Union Emissions Trading Scheme*, Cambridge University Press.

Ellerman, D. and I. Sue Wing (2003), *Absolute versus intensity-based emission caps*, Climate Policy, 3(2), S7-S20.

European Commission (2019), *Communication from the Commission to the European Parliament, the European Council, the Council, the European Economic and Social Committee and the Committee of the Regions – The European Green Deal*, COM(2019) 640 final, European Commission, Brussels.

European Commission (2020a), *Proposal for a Regulation of the European Parliament and of the Council establishing the framework for achieving climate neutrality and amending Regulation (EU) 2018/1999 (European Climate Law)*, COM(2020) 80 final, European Commission, Brussels.

European Commission (2020b), *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions – Stepping up Europe's 2030 climate ambition. Investing in a climate-neutral future for the benefit of our people*, COM(2020) 562 final, European Commission, Brussels.

Fischer, C. (2001), *Rebating environmental policy revenues: output-based allocations and tradable performance standards*, RFF Discussion Paper 01-22, Resources for the Future, Washington, D.C.

Fischer, C. (2003), *Combining rate-based and cap-and-trade emissions policies*, Energy Policy, 3S2, S89-S103.

Flachsland, C., Marschinski, R., and O. Edenhofer (2009), *To link or not to link: benefits and disadvantages of linking cap-and-trade systems*, Climate Policy, 9(4), 358-372.

Flachsland, C., Pahle, M., Burtraw, D. and O. Edenhofer, Elkerbout, M., Fischer, C., Tietjen, O. and L. Zetterberg (2020), *How to avoid history repeating itself: the case for an EU Emissions Trading System (EU ETS) price floor revisited*, Climate Policy, 20(1), 133-142.

FSR Climate (2020), *Informing the Carbon Market Policy Dialogue: the Emissions Trading Systems at a glance*, Florence School of Regulation, European University Institute, Florence, Italy.

Green, J. (2017), *Don't link carbon markets*, Nature (comment), 543(7646), 484-486.

Green, J., Sterner, T. and G. Wagner (2014), *A balance of bottom-up and top-down in linking climate policies*, Nature Climate Change, 4, 1064-1067.

Guangming, L. and W. Zhang (2017), *Research on industrial carbon emissions and emissions reduction mechanism in China's ETS*, China Population, Resources and Environment, 27(10), 141-148.

Habla, W. and R. Winkler (2018), *Strategic delegation and international permit markets: why linking may fail*, Journal of Environmental Economics and Management, 92, 244-250.

Haites, E. (2014), *Lessons learned from linking emissions trading systems: general principles and applications*, PMR Technical Note 7, Partnership for Market Readiness, World Bank.

Haites, E. and F. Mullins (2001), *Linking domestic and industry greenhouse gas emission trading systems*, Electric Power Research Institute (EPRI), International Energy Agency (IEA) and International Emissions Trading Association (IETA), Paris.

Helm, C. (2003), *International emissions trading with endogenous allowance choices*, Journal of Public Economics, 87(12), 2737-2747.

Hepburn, C., Neuhoff, K., Acworth, W., Burtraw, D. and F. Jotzo (2016), *The economics of the EU ETS markets stability reserve*, Journal of Environmental Economics and Management, 80, 1–5.

Hintermann, B., Peterson, S. and W. Rickels (2016), *Price and market behavior in Phase II of the EU ETS: a review of the literature*, Review of Environmental Economics and Policy, 10(1), 108-128.

Holtsmark, K. and K. Midtømme (2019), *The dynamics of linking permit markets*, CESifo Working Papers No. 7548, Munich.

Holtsmark, B. and D.E. Sommervoll (2012), *International emissions trading: good or bad?* Economics Letters, 117(1), 362-364.

Holtsmark, B. and M. Weitzman (2020), *On the effects of linking cap-and-trade systems for CO_2 emissions*, Environmental and Resource Economics, 75, 615-630.

ICAP (2020), *Emissions trading worldwide – Status report 2020*, International Carbon Action Partnership, Berlin.

Jaffe, J and R.N. Stavins (2008), *Linkage of tradable permit systems in international climate policy architecture*, NBER Working Paper 14432, National Bureau of Economic Research, Cambridge, Massachusetts.

Klemetsen, M., Rosendahl, K.E. and A.L. Jacobsen (2020), *The impacts of the EU ETS on Norwegian plants' environmental and economic performance*, Climate Change Economics, 11(1), 2050006.

Lapan, H. and S. Sikdar (2019), *Is trade in permits good for the environment?*, Environmental and Resource Economics, 72(2), 501-510.

Lazarus, M., Schneider, L., Lee, C. and H. van Asselt (2015), *Options and issues for restricted linking of emissions trading systems*, ICAP report, International Carbon Action Partnership, Berlin, Germany.

Marschinski, R. (2008), *Efficiency of emissions trading between systems with absolute and intensity targets*, Paper presented at the EAERE 2008 Annual Conference (Gothenburg).

Mehling, M., Metcalf, G. and R. Stavins (2018), *Linking climate policies to advance global mitigation*, Science, 359(6379), 997-998.

Osorio, S., Tietjen, O., Pahle, M., Pietzcker, R. and O. Edenhofer (2020), *Reviewing the Market Stability Reserve in light of more ambitious EU ETS emission targets*, ZBW – Leibniz Information Centre for Economics, Kiel, Hamburg

Perino, G. and M. Willner (2017), *EU-ETS Phase IV: allowance prices, design choices and the Market Stability Reserve*, Climate Policy, 17(7), 936-946.

Pizer, W. and A. Yates (2015), *Terminating links between emission trading programs*, Journal of Environmental Economics and Management, 71, 142-159.

Quemin, S. (2020), *Using supply-side policies to raise ambition: the case of the EU ETS and the 2021 review*, CCCEP Working Paper 366, Centre for Climate Change Economics and Policy.

Ranson, M. and R. Stavins (2016), *Linkage of greenhouse gas emissions trading systems: learning from experience*, Climate Policy, 16(3), 284-300.

Rehdanz, K. and R. Tol (2005), *Unilateral regulation of bilateral trade in greenhouse gas emission permits*, Ecological Economics, 54(5), 397-416.

Sterk, W., Braun, M., Haug, C., Korytarova, K., and A. Scholten (2006), *Ready to link up? Implications of design differences for linking domestic emissions trading schemes*, German Federal Ministry of Education and Research (BMBF), Jet-Set Cross-Section Project 4.

Sue Wing, I., Ellerman, D. and J. Song (2008), *Absolute versus intensity limits for CO2 emission control: performance under uncertainty*, In: Tulkens, H and R. Guesnerie (eds.), *The design of climate policy*, MIT Press, 221-252.

Wang, W., Xie, P., Li, C., Luo, Z. and D. Zhao (2018), *The key elements analysis from the mitigation effectiveness assessment of Chinese pilots carbon emission trading system*, China Population Resources and Environment, 28, 26-34.

Yi, L., Bai, N., Yang, L., Li, Z. and F. Wang (2020), *Evaluation on the effectiveness of China's pilot carbon market policy*, Journal of Cleaner Production, 246, 119039.

Zetterberg, L. (2012), *Linking the emissions trading system in EU and California*, Stockholm, S.E., Swedish Environmental Research Institute.

Zhang, H., Duan, M. and Z. Deng (2019), *Have China's pilot emissions trading schemes promoted carbon emission reductions? The evidence from industrial sub-sectors at the provincial level*, Journal of Cleaner Production, 234, 912-924.

Zhang, W., Li, J., Li, G. and S. Guo (2020), *Emission reduction effect and carbon market efficiency of carbon emissions trading policy in China*, Energy, 196, 117117.

QM-01-20-699-EN-N

Publications Office

doi:10.2870/801301
ISBN:978-92-9084-936-0