CHAPTER 23

The Trading of Carbon

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1 Introduction

The rationale for establishing carbon markets is based on the economic theory of market externalities. This theory was initiated by the work of R. H. Coase in 1960, thanks to an article titled 'The Problem of Social Cost', which highlighted the inefficiencies caused by market externalities (Coase 1960). Since then, the theory of externalities has analysed circumstances when actions of one economic agent make another economic agent worse or better off, yet the first agent neither bears the costs nor receives the benefits of doing so (Saez 2007). When one's actions have harmful effects on the others, this is called a 'negative externality' as opposed to a 'positive externality'.

One case in point of negative externalities is carbon emissions, as the emitter can 'externalise' the cost associated with its emissions by passing it on to society. This negative externality is a market failure, because the emitter does not bear the true costs of emitting.

To respond to the market failure of carbon emissions, two types of approaches can be envisioned, the two not being mutually exclusive:

- Command and control measures
- Economic instruments

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M. Hafner, G. Luciani (eds.), *The Palgrave Handbook of International Energy Economics*, https://doi.org/10.1007/978-3-030-86884-0_23

A command and control approach means that a host of rules and restrictions are imposed on producers and consumers, with the aim of constraining emissions. Accordingly, certain behaviours or production methods might be disincentivised, or prohibited altogether.[1]

On the other hand, approaching carbon emissions through economic instruments means that a price signal is associated with GHG emissions. This signal could be either positive, incentivising certain activities or behaviours (e.g. by granting subsidies), or negative, imposing a cost on emissions.

The idea that polluters should be charged for their emissions according to the corresponding cost caused to society by their actions reflects the so-called polluter pays principle (LSEPS 2018a). This principle translates in applying a cost to carbon pollution, giving an economic signal to emitters who will decide for themselves whether to reduce the amount of greenhouse gases they emit or continue polluting yet pay for it (World Bank 2019). In essence, emitters will be forced to 'internalise' the cost of pollution, ending the 'negative externality' of carbon emissions.

Countries have applied both command and control and economic approaches to the issue of climate change. In the quest to limit global warming well below 2°C above pre-industrial levels, in line with the Paris Agreement, the two methods should be seen as complementary.

Looking at the potential economic approaches to carbon emissions, economists widely agree that introducing a carbon price allows to achieve environmental goals in the most flexible and cost-effective way to society (LSEPS 2018b).

This chapter will explore the concept of carbon pricing, with a specific focus on carbon trading via emissions trading systems (ETSs). The first section will analyse the rationale for ETSs as opposed to carbon taxes, looking at the main differences between these two approaches to carbon pricing. The second section will highlight the main design options for a cap-and-trade system.

The third section will focus on the experience of the European Union (EU) ETS, being the world's first international ETS and still today the biggest one. The history of the EU ETS will be examined, explaining what the main challenges and benefits of the system are, and what lessons can be learnt from this ongoing experience.

The fourth section will broaden the analysis to other existing ETSs worldwide, while also trying to shed light on the potential for international cooperation in the area of carbon trading. Using the EU ETS as a benchmark, other major ETSs will be briefly described, highlighting the potential for interconnection of different systems, as well as analysing the outlook for the emergence of a reference global carbon price.

[1] This was the case of the Montreal Protocol, which was created to eliminate the production of chlorofluorocarbons (CFCs) that were found to contribute to a hole in the earth's ozone layer. To clarify further, see: Heskett (2018).

2 Pricing Carbon: Carbon Taxes Versus Emissions Trading Systems

Carbon pricing is gaining momentum worldwide. As of 2019, 57 carbon pricing initiatives have been implemented or are scheduled for implementation, with 46 national and 28 subnational jurisdictions placing a price on carbon emissions through a combination of ETSs and taxes. This equals to roughly 20% of global emissions, up from 13% in 2016 (ICEVE 2019).

Against this backdrop, studying how carbon pricing works is becoming increasingly important. There are two main alternatives to deliver an explicit price on carbon:

- Carbon tax
- Emissions trading system

In broad terms, a carbon tax implies that the price for carbon emissions is fixed by regulatory authorities. An emissions trading system fixes the total amount of emissions allowed under the system (i.e. a cap), and leaves it to the market to find the price needed to constrain emissions within that limit.

The key difference between these two alternatives is which variable the regulator chooses to make predictable: the carbon price or the total emissions under the scheme (Stenegren 2018). In the case of a carbon tax, there is certainty on the carbon price as set by the government, whereas the quantity of emissions is not controlled. Entities are allowed to emit as long as they pay for their emissions, with the carbon tax acting as an economic disincentive towards emitting emissions.

Conversely, an ETS creates certainty on the environmental outcome by fixing a cap on the quantity of emissions.[2] However, the price is determined by the market according to the supply-demand balance of emission 'allowances' (i.e. emissions permits). Therefore, the price of carbon is not fixed, in principle, but fluctuates according to market conditions.

Hybrid systems, combining elements of both approaches, also exist in different forms. One example is an ETS with a price floor and/or ceiling (ICAP 2016).

Both approaches share several advantages over alternative policies (Kaufman et al. 2016). Firstly, both carbon taxes and ETSs reduce emissions by encouraging the lowest cost emissions reductions, without dictating where and when the emissions reduction should take place. Secondly, they both represent an economic incentive to low-carbon investments and can therefore help driving technological innovations. Third, both policies generate revenues for the

[2] Unless the legislator chooses an intensity cap for the ETS, which would guarantee the achievement of an intensity target but not of an absolute emissions reduction target. This is the case of the Low Carbon Fuel Standard in California. To clarify further, see: https://ww3.arb.ca.gov/fuels/lcfs/background/basics-notes.pdf

government, provided that allowances under an ETS are auctioned and not handed out for free.

At the same time, any ETS or carbon tax at the sub-global level runs the risk of harming the competitiveness of the economic actors covered. This is usually referred to as the risk of 'carbon leakage'. Companies in countries without or with less stringent emissions constraints could be advantaged over the regulated entities to a 'tight' ETS or a 'high' carbon tax, as the latter will be required to bear higher carbon costs.

This loss of competitiveness could lead to the perverse effect of reducing emissions in one jurisdiction yet increasing them somewhere else, with total global emissions potentially increasing. Indeed, production could shift to countries with laxer emissions constraints, through businesses moving or being outcompeted by foreign competitors not facing similar climate-related costs (European Commission 2019a). This is especially the case for 'trade-exposed' products (Tiche et al. 2014).

Another common challenge for both approaches lies in the complex administrative structure they require, as monitoring (M), reporting (R), and verifying (V) emissions can be particularly challenging, especially in the context of countries with limited administrative capacities.

Carbon taxes and ETSs also come with specific positive and negative sides. As mentioned before, by not putting a limit on emissions, a carbon tax runs the risk of not reaching environmental targets. Setting a carbon tax might require adjustments according to the elasticity of demand of companies.

Furthermore, carbon taxes are oftentimes associated with the negative 'stigma' of being an additional tax for taxpayers, making it usually difficult to secure public support—unless a carbon tax is directly linked to tax rebates or revenue recycling. On the positive side, the design of a carbon tax is generally easier than a cap-and-trade system, and it provides investors with a stable price signal.[3]

Ensuring a long-term, explicit carbon price is particularly important, as it guarantees a long-term signal for investments in low-carbon practices and technologies, even at times when emissions reductions might become cheaper than expected or are caused by other causes—as in the case of an economic downturn causing emissions to temporary fall, regardless of the actual emission abatement efforts.

On the other hand, a cap-and-trade system has the advantage of ensuring delivery of a predetermined emissions target, providing a reasonable confidence about the future level of emissions. This emissions reduction will take place progressively, given that an ETS typically decreases the supply of allowances over time by lowering the cap.

[3] The design of a carbon tax could lead to an equally complex system, according to how benchmarks and threshold for applying the tax are designed, but an ETS is generally perceived as being more complex given that it also includes trading operations.

ETSs also offer the great advantage of a higher degree of flexibility compared to a carbon tax in terms of abatement opportunities, as it allows actors to pay others to reduce their emissions first, if they have more cost-effective alternatives to decarbonise—or, put differently, actors with lower abatement costs have the possibility to get rewarded by others for reducing their emissions first (Eden et al. 2016).

One key downside of an ETS without a price floor is that it cannot guarantee a long-term, explicit price signal the way a carbon tax would do, considering that the price of carbon will be determined by supply-demand trends.

Leaving aside any specific consideration about the pros and cons of the two alternatives, it is worth highlighting how both can deliver positive environmental results, as long as they are well designed. The following section analyses in more depth what are some of the main design options for an emissions trading system.

3 Design Options for Emissions Trading Systems

There are many design options that can be considered for an ETS, including provisions mimicking the behaviour of a market with flexibility on both supply and demand sides. This section will focus on the following seven points, given their relevance for an ETS functioning[4]:

- Cap setting
- Scope and coverage
- Supply of allowances to the market
- Flexibility provisions
- MRV, enforcement and market oversight
- Price and quantity management mechanisms
- Use of revenues

3.1 Cap Setting

The cap represents one of the most important design characteristics of an ETS, as it defines the upper limit of GHG emissions allowed under the system. If an ETS aims at reaching a given environmental target of cutting emissions by X% over a set period of time, the legislator will set a ceiling on emissions (so-called emissions budget) equal to the total number of allowances available to covered entities over that period (European Commission 2015). This cap has to be translated into units with a face value, which are used by covered installations for trade and compliance—usually referred to as emissions allowances or emissions permits.

[4] Section 3 is by and large based on: Laing and Mehling (2013), Newell et al. (2012), Prag et al. (2012), Neuhoff (2008), ICAP (2019a)

The cap should reflect the mitigation opportunities available, the economic and technological feasibility of meeting the target for covered entities, but primarily the jurisdiction's overall mitigation objectives. In this sense, having a robust foundation of both historical data and counterfactual projections is a key pre-requisite to the cap setting.

When setting the cap, the legislator should seek to balance the reasons for emissions reduction with the economic implication of a 'tighter' cap for covered entities. All else being equal, the more ambitious the cap the lower the amount of allowances issued is, and thus the greater the scarcity in the market, which will translate into higher compliance costs.

Two types of cap can be set:

- an absolute cap, setting the maximum quantity of emissions allowances over a given period of time;
- an intensity cap, setting the number of allowances issued per unit of output or input (e.g. emissions per unit of GDP, kilowatt-hour of electricity, tonne of raw material, etc.).

An absolute cap has the benefit of providing upfront certainty to both regulators and market participants, being an independent variable that does not change according to other factors. On the contrary, under an intensity approach, the absolute amount of emissions allowed under the cap will increase or decrease as a function of the input or output chosen. In both cases, the legislator will need to choose a reference point for emissions.

Another distinction is the time period considered for the cap, which can be annual or on a multiyear basis. This typically means choosing a starting date and an end date.

In practice, the cap period usually corresponds to a commitment period or 'ETS phase', during which other programme design features are also specified. Decisions on the duration of the ETS phase(s) will influence the flexibility given to stakeholders to decreasing their emissions throughout time.

Ultimately, two approaches are available when deciding on the ambition of an ETS cap: top-down and bottom-up. In the case of the former, the legislator will set the cap in line with the underlying environmental target to be met, ideally reflecting a high-level assessment of the mitigation potential and costs across all capped sectors.

For the latter, a bottom-up approach will translate in a more detailed assessment of the mitigation potential of the different sectors, subsectors or even individual participants, and aggregating that into an overall cap. A bottom-up approach has the advantage of reflecting the specificities of all or most entities that are part of an ETS. However, it requires high-quality, disaggregated data, and might still result in the cap not being aligned with the broader mitigation target of the jurisdiction.

3.2 Scope and Coverage

The scope and coverage of an ETS refer to the geographical area, sectors, emissions sources and gases that are covered under an ETS. It defines the boundaries of the policy, and will have critical repercussions on the administrative efforts and transaction costs for regulated entities. A broad number of sectors and gases covered would, in theory, increase the opportunities for low-cost abatement, reducing the overall marginal costs for emission reductions while increasing liquidity in the regulatory market created.

On the other hand, limiting the scope and coverage lowers the bureaucratic costs related to an ETS, both for the legislator and for the regulated entities. Of course, this is to be balanced against the costs of covering sectors/entities outside of the ETS with other climate regulations. The ability and cost for the legislator of monitoring and regulating a large number of actors and emissions sources should not be dismissed as marginal in the decision of an ETS scope, especially if alternative policies might also be envisioned.

The legislator should seek to limit compliance costs for all covered entities. This is especially true for those small entities who would not be able to bear high administrative costs, and who might suffer a competitive disadvantage when asked to cover similar fixed administrative burdens to that of major emitters. For these reasons, ETSs will usually aim at covering major emitters, creating some thresholds under which smaller entities will be exempted from compliance. These thresholds can reflect both economic and environmental considerations, such as production levels or GHG emissions per year.

Similar considerations apply to the decision of which sectors should be included in an ETS. Covering sectors representing a big share of a jurisdiction's emissions will be seen as more beneficial towards achieving the ETS environmental goal, whereas covering less pollutant sectors could sometimes be considered not worth the administrative costs, if, an alternative policy could be implemented.

Similarly, sectors dominated by a small number of large emitters can provide high benefits as well as limit administrative efforts, whereas covering sectors composed of many small entities may involve disproportionately high costs relative to benefits.

Every decision on the scope of an ETS might have repercussions on market liquidity and market power, as well as on the competitiveness of different actors. Creating exemptions for entities or sectors always has the potential for these repercussions.

An additional design feature concerns the point at which those emissions are regulated. For a number of emissions sources, especially those involving fossil fuel use, the main 'points of regulation' are:

- upstream, that is, where the source of emissions is first commercialised, or where non-energy process emissions are generated from industrial activities;
- downstream, that is, where GHG emissions are physically released into the atmosphere.[5]

Regulating emissions upstream has the advantage of lowering the administrative costs, considering that there are usually fewer entities involved in extraction and commercialisation of carbon-intensive resources, as opposed to the number of actors using those resources as final consumers. Furthermore, the coverage across sectors tends to be higher with an upstream regulation: regulating emissions upstream reduces the need for sectoral thresholds compared to downstream systems, where thresholds are usually required to avoid high transaction costs.

On the other hand, downstream regulation might be preferred if installation-level data of downstream users already exists, limiting the administrative burden of regulating emissions at the point where they are actually released (ICAP 2016). Many existing ETSs, including the largest EU ETS, have a downstream point of regulation.

In terms of the gases covered, all the existing ETSs include carbon dioxide as a minimum. Some ETSs include other gases, such as methane, N_2O, PFCs, and so on. If an ETS covers GHGs other than CO_2, their emissions are typically expressed as carbon dioxide equivalent (CO2e), to facilitate the measuring and trading of allowances.

3.3 Supply of Allowances to the Market

Allocation is the process of distributing emission allowances to covered entities in an ETS. Given that these allowances have an associated value, the way they are distributed should reflect considerations of fairness and cost distribution across the society.

Allowances can be either sold through auctions or distributed for free according to pre-set rules (so-called free allocation). For auctioned allowances, their selling generates an income stream for the state, and the price at which allowances are sold reflects their perceived scarcity in the market, presumably connected to secondary market price.

Apart from generating revenues, selling allowances through auctions has the advantage of reflecting the actual demand on the ETS. This facilitates price discovery and market liquidity. Furthermore, auctioning clearly reflects the 'polluter pays' principle.

However, auctioning also has disadvantages, particularly as it does not offer protection against the risk of carbon leakage and of loss competitiveness. To

[5] This also includes non-energy process emissions generated from industrial activities.

avoid such risks—especially carbon leakage—free allocation of allowances has oftentimes been seen as necessary in a period of transition.

There are two main methods for handing out allowances for free:

- grandfathering
- benchmarking

Through grandfathering, regulated entities receive permits according to historical emissions during a given period of time. This approach limits initial costs for covered sectors but runs the risk of benefiting historically high emitters with windfall profits, as allocation is not linked to the actual performance or production during the ETS periods. New entrants will also be disadvantaged, unless specific provisions are put in place.

When using benchmarking regulated entities receive free allocation according to some performance indicators. This can translate in sector benchmarks or output-based allocation (OBA), aiming to reward efficient installations and early movers that actively embark in emissions reduction strategies.

The calculation of benchmarks will depend on the availability of reliable and robust data collection. Only perfectly accurate data will ensure the same economic efficiency as auctioning allowances. Otherwise, any type of free allocation will, in practice, be less economically efficient than auctioning, with risks of over- or under-allocation. This risk is also dependent on production levels.

3.4 Flexibility Provisions

In the analysis of the cap setting it was mentioned how the decision on the time period for setting the ETS phase(s) has implications on the temporal flexibility provided to stakeholders. In essence, temporal flexibility refers to when or how quickly regulated entities need to achieve their emissions reductions.

Some provisions can be designed to increase an ETS temporal flexibility, including:

- banking of allowances from the current compliance period for use in future periods;
- borrowing of allowances from future compliance periods to the current period.

The banking of allowances allows regulated entities to build a buffer against future high prices, if they consider early-on mitigation options as less expensive than in future compliance periods. Borrowing provisions provides entities with higher flexibility in the determination of their compliance strategy, particularly in those sectors where abatement opportunities take longer to bear fruits.

Both borrowing and banking provisions, however, might have negative effects on the functioning of an ETS. Unlimited banking, for instance, could carry forward the effects of an oversupply of permits, whereas allowing very generous borrowing could give companies an incentive to delay emissions reduction indefinitely (ICAP 2017).

A different set of flexibility provisions includes the use of mitigation offsets and/or allowances from a baseline and credit system. In addition, linking with other carbon markets is also a potential source of flexibility for covered entities. These are usually referred to as 'geographical flexibility' (ICAP 2017).

The main reason for opening up an ETS to domestic and/or international offsets, or linking an ETS with other cap-and-trade systems, is to reduce the overall costs for compliance and increase liquidity. Theoretically, by expanding the compliance opportunities, regulated entities should be able to seek the most cost-effective abatement option across a wider range of opportunities.

Domestic offsets provide credits for emissions reductions taking place in sectors non-covered by an ETS, yet within the same jurisdiction. International offsets give credits for mitigation actions taking place outside the ETS-jurisdiction, provided that these actions comply with some pre-set standards of environmental integrity.[6]

Additional flexibility can be provided through the 'linking' of two or more ETSs. If two systems mutually recognise their respective allowances for domestic compliance (full linking), this should offer participants with a larger carbon market within which to operate. As an alternative, a unilateral direct linking implies that a given ETS A explicitly recognises allowances from ETS B as eligible for compliance, but not vice versa (Borghesi et al. 2016).

As in the case of domestic and international offsets, linking should increase the compliance flexibility for the stakeholders of an ETS, by expanding the range of abatement options available and, likely, lowering the average cost of allowances of both ETSs.

At the same time, any kind of 'geographical flexibility' mechanism might also cause problems to an ETS, notably in terms of supply-demand balance and potential for carbon lock-ins. Jurisdictions usually limit the number of offsets that may be used, to ensure that most of the abatement efforts take place within the domestic ETS (Borghesi et al. 2016).

3.5 MRV, Enforcement and Market Oversight

For any ETS to function properly, control and enforcement measures are very important. There has to be a high level of trust that emissions are accurately monitored (M), reported to regulators (R), and verified (V); that market oversight is guaranteed; and that non-compliance is effectively sanctioned.

Most ETSs have established legal MRV frameworks to track compliance and guarantee the principle that a 'ton is always a ton'.[7] Accordingly, emissions are

[6] Historically, examples of international credits are credits generated by Joint Implementation projects and Clean Development Mechanism project activities under the Kyoto Protocol.

[7] This principle implies that a claimed emissions reduction of X tonnes of CO2 reflects an equivalent atmospheric reduction.

measured either via direct monitoring (real-time emissions) or using emission factors of fuels or chemical processes.

Measured emissions are reported to the relevant authority—usually government inspectors or third-party experts—which will be responsible for the auditing and verification of the compliance of regulated entities (Wettestad and Gulbrandsen 2017). Registries—databases that record and monitor the creation, trading, and surrender of all units within a system—also need to be developed and verified (ICAP 2016).

The impartiality and fairness of the verification mechanisms are key to guarantee the trustworthiness of the system. In cases of non-compliance, enforcement provisions have to identify penalties. Enforcement provisions can include monetary sanctions as well as criminal penalties.

Finally, a cap-and-trade system should ensure that the trade of permits is not vulnerable to fraud and manipulation. Market oversight provisions are therefore needed, in order to safeguard the integrity of the trading activities. Such provisions should seek to facilitate price discovery by increasing transparency, containing risk, maximising liquidity, and ensuring fair competition (Kachi and Frerk 2013). Market oversight should apply equally to primary and secondary markets, as well as to derivatives contracts.

3.6 Price/Quantity Management Mechanisms

ETSs are regulatory markets and, as any other market, can incur in supply-demand imbalances for a multiplicity of reasons. Endogenous or exogenous shocks, regulatory uncertainty, and the existence of flaws in the design of the market are just a few examples of reasons for a carbon market to not live up to its expected performance.

In this context, the legislator may want to intervene in the market to avoid excessive price variability, ensure cost containment, and/or improve the resilience of an ETS from the effects of different types of shocks (e.g. economic events, policy overlaps, etc.). Two market interventions that can be envisioned include:

- price management mechanisms
- quantity management mechanisms

Price mechanisms usually aim at limiting excessive price volatility, while ensuring medium- to long-term stability of the price signal to the market. Approaches in this sense include the creation of a price ceiling to avoid excessively high prices, a price floor to guarantee that prices do not fall below a certain threshold, or a price corridor having both a floor and a ceiling.

The rationale for quantity-based mechanism is similar, in the sense that they are implemented to improve the supply-demand balance of an ETS. However, quantity-based mechanisms act on the allowance volumes, avoiding lengthy political debates on the establishment of price thresholds. Approaches to quantity management mechanisms include the creation of a quantity-triggered reserve, adjusting the supply of allowances of an ETS according to some predefined triggers. These triggers do, however, need a political agreement determining their functioning, including what thresholds trigger the quantity-based mechanisms.

Both price and quantity management mechanisms should be carefully designed, keeping in mind that market interventions can also lead to market distortions. Furthermore, any legislative intervention has the potential to increase regulatory uncertainty, thus decreasing the overall confidence in the system.

3.7 Use of Revenues

Governments can use ETS-revenues in multiple ways, from adding those revenues to the general budget, to earmarking carbon revenues for specific purposes. In evaluating options, there are several key principles a jurisdiction should consider, including: potential economic and environmental gains, efficiency, interaction of spending with the carbon price itself (Carbon Pricing Leadership Coalition 2016).

One possible approach is to use revenues for public spending, as part of a jurisdiction's general budget. This could translate in financing other tax reductions, cutting the public debt and/or deficit, or, more broadly, using carbon revenues for expenditures not related to the ETS. This is perfectly justifiable from an economic standpoint, as economists often consider earmarking as being inefficient (Carbon Pricing Leadership Coalition 2016).

However, empirical evidence shows that ETS-revenues are often channelled to further climate action or to compensate particularly vulnerable groups. The main rationale for this is to increase the political and social acceptability of putting a price on carbon, and increase the environmental delivery of an ETS—thus managing the transition (Santikarn et al. 2019).

Actions to compensate vulnerable groups include addressing fairness and competitiveness concerns arising from the ETS, for example by directing revenues to low-income households suffering the effects of ETS prices or supporting industries at risk of carbon leakage.

Examples of earmarking ETS-revenues for climate action include supporting investments in low-carbon technologies and innovation, financing climate and energy programmes, and/or incentivising adaptation strategies to limit climate change impacts (Santikarn et al. 2019).

4 The Experience of the EU ETS and its Ongoing Evolution

4.1 The Early Years of the EU ETS

Since 2005, the EU ETS has been a core element of the EU's policy to combat climate change, and, according to the European Commission, a 'key tool for reducing greenhouse gas emissions cost-effectively' (European Commission 2019b).

In more than ten years of operations, the system had to overcome ad-hoc challenges, and went through different reforms and changes. Analysing the history and learning process of first major ETS can be very useful to understand some of the key lessons that can be extrapolated from the European experience. These lessons could help others in developing their respective ETSs.

Starting from the beginning, it is important to appreciate that the advent of the EU ETS came as a result of the failure to achieve a political agreement on the implementation of a carbon tax in Europe during the 1990s. In the aftermath of the adoption of the 1992 United Nations Framework Convention on Climate Change (UNFCCC), the European Commission (EC) started an internal debate to promote an EU-wide carbon tax to tackle Europe's GHG emissions (Andersen and Ekins 2009).

However, this proposition found the opposition of many industrial players and a group of EU Member States (MS) led by the United Kingdom, who portrayed the idea of a CO_2/energy tax as largely detrimental for Europe's international competitiveness (Dupont and Moore 2019). This opposition also stemmed from political considerations, with some MS being unwilling to give away legislative powers to the EC on taxation measures, considering that taxation is a core competence of EU MS (Climate Policy Info Hub 2019a).

It was only after the introduction of the Kyoto Protocol in 1997 that the idea of pricing carbon came back at the centre of the EU policy-making debate. The Kyoto Protocol acted as the enabling framework within which the EU ETS came to life. Article 17 mentioned that the parties to the Protocol may engage in emissions trading to achieve their emissions reduction targets, adding that any such trading should be 'supplemental to domestic actions' (UNFCCC 1998).

Article 17 helped the creation of the EU ETS in two ways. First, it created a new commodity which could be traded internationally, shaping the idea that countries could 'trade carbon' in the form of assigned amount units (AAUs), that is, the unit used to define the Kyoto Protocol GHG emission targets.

Second, it promoted a discussion on the 'supplementarity' of international emissions trading. This debate discouraged the idea that rich countries could just buy off allowances without engaging in domestic mitigation actions (Albrecht 2002). Facing with the perspective that the EU would have to reduce its domestic emissions one way or another, the suspicions of the European business community towards a EU-wide carbon market started to fade away (Mäenpää 2016).

Moreover, the idea of implementing an ETS in Europe had the political advantage of it not being a tax. The EC managed to present the EU ETS as an environmental measure, which would not need a unanimity vote within the European Council. This helped alleviate the opposition of those MS who had opposed to a carbon tax in the early 1990s (Climate Policy Info Hub 2019b).

Furthermore, at that point in time there was already one pioneering example of a cap-and-trade system in the field of environmental regulation: the sulphur dioxide (SO_2) emissions trading system in the United States. This market was engineered by Richard L. Sandor, and addressed the threat of acid rain as part of the Clean Air Act Amendments of 1990 (Chan et al. 2012). The SO_2 market proved to be very successful in reducing emissions in a timely and cost-effective manner, showing in practice that market mechanisms could help achieving environmental goals (Sandor 2016). The design and experience of the SO_2 market became a reference for policy-makers and researchers in favour of cap-and-trade systems for decades to come.

It was in this context that in the late 1990s the EC started elaborating a proposal for an EU ETS addressing emissions from key economic sectors, presenting it as a key measure to reach the Kyoto Protocol targets. The legitimacy the EU ETS was given through the Kyoto Protocol was important in securing the support of the European environmental community, which was initially sceptical about the proposal, but wanted the Kyoto Protocol to succeed. The support of the EU environmental groups was further reinforced by the idea that the EU ETS would contain a binding cap, and the cap would decrease throughout time.

The first two phases of the EU ETS were agreed upon by the EU institutions in 2003. This first phase was designed as a pilot phase for the period 2005–2007, with the idea that regulated entities could use it as a 'learning by doing', yet not creating any continuity with the following trading periods through mechanisms of intertemporal flexibility (e.g. borrowing, banking).

The allocation of European Union Allowances (EUAs) was done through national allocation plans (NAPs) via free allocation based on grandfathering.[8] The main rationale for NAPs was to secure MS support towards the system, leaving them with the task to set their own cap and determine the distribution of allowances to affected facilities, subject to a review by the EC. As a result of this, the EU-wide cap was in practice the sum of MS caps (Ellerman et al. 2015).

The point of regulation was set downstream. In Phase 1, the scope of the EU ETS included both the power sector and large industrial installations (Zetterberg et al. 2014). This meant that over two billions tCO2e emissions were covered in 2005, amounting to ca. 37.6% of total EU emissions (EEA 2019).

However, the allocation of EUAs was too generous to ensure the supply-demand balance in the market. By grandfathering allowances based on historic production levels, the allocation did not reflect actual emissions of the covered

[8] The fact that MS were left with the task of distributing allowances during the EU ETS Phase 1 and 2 helped securing political support towards the system.

installations. Moreover, MS did not want to risk jeopardising the competitiveness of their national businesses, and tended to over-allocate allowances through NAPs.

As soon as it became evident that the Phase 1 supply would outstrip demand, prices started decreasing, to end up at zero by the end of Phase 1, reflecting the absence of borrowing/banking mechanisms.

Nevertheless, Phase 1 was still beneficial on many aspects:

- first, it built a framework for MRV;
- second, it encouraged data collection for installations' emissions, which would later be used as a baseline;
- third, it made stakeholders trade carbon for the first time, creating a price for carbon in Europe.

During the first three years of the EU ETS operations, around 200 million tonnes of CO_2, or 3% of total verified emissions, were abated (IETA 2011).

4.2 Challenges and Reforms: EU ETS Phase 2 and Phase 3

The second phase of the EU ETS corresponded to the first commitment period under the Kyoto Protocol (2008–2012) and saw the EU adopting the goal of reducing emissions under the EU ETS cap by 6.5% compared to 2005 levels. Some of the initial key design elements included (Delbeke and Vis 2016):

- a binding, enforceable and decreasing cap placed on absolute GHG emissions;
- a percentage of allowances to be auctioned for the first time (up to 10%);
- continuation of the national allocation plans (NAPs), the sum of which established the overall cap of the system (European Commission 2019c, 2019d);
- introduction of both borrowing and banking mechanisms;
- acceptance of international credits from the Kyoto Protocol's clean development mechanism (CDM) and joint implementation (JI) projects, which could be used for compliance on top of the EU ETS cap (up to 1.4 billion tonnes of CO2e).
- introduction of intra-EU flights into the system, as of January 1, 2012;
- penalty for non-compliance rising from 40 to 100 €/tCO2e, compared to Phase 1.

Furthermore, from the beginning of Phase 2, Iceland, Norway and Liechtenstein also linked their domestic ETSs with the EU ETS (European Commission 2007).

Phase 2 marked the actual start of the European carbon market, but it took less than one year of operations for the system to suffer its first major crisis. Indeed, as soon as the 2008 economic crisis hit the European economy, the demand for allowances dropped, leading to the price of EUAs going from

30€/tCO2e in June 2008 to less than ca. 9€/tCO2e in February 2009 (Sandbag 2019). This revealed the fundamental weakness of the EU ETS design, putting the system's credibility in serious jeopardy (Climate Policy Info Hub 2019a).

The combination of decreasing EUAs demand and stable supply based on grandfathering on production levels led to the built-up of a significant surplus of allowances, with companies receiving more EUAs than the amount they needed for compliance. The creation of the surplus was further exacerbated by the availability of cheap international credits coming to the market on top of the EU ETS cap, which contributed to maintaining prices well below 10€/tCO2e for several consecutive years.

To make things worse, MS had a perverse incentive to allocate EUAs as generously as possible to their industries via NAPs, as failing to do so would put neighbouring countries' actors at a competitive advantage, particularly in the context of a concurrent macroeconomic recession (Stenegren 2018).

The system design had a lack of flexibility of supply, both in free allocation and auctioning. The EU ETS was not able to react to supply and/or demand shocks. On the supply side, the low cost of international credits led to high imports of credits—increasing supply as the number of EUAs allocated and auctioned was not adjustable. On the demand side, the 2008 economic crisis on the demand side led to significant reductions to GHG emissions through slowing economic activity—including GHG emitting ones. Carbon prices ended dropping, and becoming too low to drive investments in emissions abatement. Additionally, the effects of other EU-wide overlapping policies (e.g. legislations aiming to promote energy efficiency and renewable energy sources) also contributed to further decrease the demand for EUAs.

In order to address the supply-demand imbalance and restore confidence in the system, the EC promoted a first reform of the EU ETS in 2009, the effects of which would unfold in the EU ETS Phase 3 (2013–2020) (European Parliament and Council 2009).

Compared to Phase 2, an EU-wide cap on emissions was set centrally replacing the system of NAPs. This cap reflected the EU's ambition to cut emissions in EU ETS sectors by 21% by 2020. To achieve this target, the cap was designed to decrease each year by a linear reduction factor (LRF) of 1.74% of the average total quantity of allowances issued annually in 2008–2012 (European Commission 2019e). Other new features of Phase 3 were:

- a longer commitment period of 7 years;
- auctioning partially replaced free allocation, and sectoral benchmarks were used for the remaining free allocation, instead of grandfathering (European Commission 2019f)[9];

[9] Generally speaking, a product benchmark under the EU ETS is based on the average GHG emissions of the best performing 10% of the installations producing that product in the EU.

- more stringent qualitative and quantitative limits on the imports of international credits[10];
- increase of scope;
- requirement for MS to spend at least 50% of the ETS-revenues in support of the achievement of specific climate and energy activities;
- establishment of the NER300, a funding programme financed through ETS-revenues, supporting innovative low-carbon energy demonstration projects.

Under the Phase 3 reform, the power sector was entirely moved to auctioning (with the exception of some solidarity clauses for lower income MS), while industrial installations considered at significant risk of carbon leakage continued to receive free allocation through benchmarks (Stenegren 2018). This was done to limit windfall profits, as the electricity sector is able to pass ETS additional costs through to customers, as it is not subject to international competition.

However, these reforms did not address the main problem of the EU ETS, namely the accumulated surplus carried on from the early years of the EU ETS, which continued to hamper the EU ETS supply-demand balance.

The short-term legislative response to addressing the surplus was to reduce the quantity of allowances available for auctioning in the years 2014 to 2016 by 900 million allowances, and re-inject them into the market in the year 2019–2020 (Delbeke and Vis 2016). This 'back-loading' measure was agreed on by the European institutions in 2011 as a stop-gap measure and helped containing the oversupply of EUAs at least in the short term (EU Commission Regulation 2011).

However, the reform did not eliminate the surplus in the medium to long run, as all the backloaded allowances were expected to come back to the market in 2019–2020. This would have caused a rebound of the surplus, albeit a few years later.

Facing with this situation, the EU institutions were left with the task of adopting a more comprehensive reform to the system, which took place via two separate legislations:

- the Market Stability Reserve (MSR) Decision, adopted in 2015 (European Parliament and Council 2015);
- the revised EU ETS Directive for Phase 4 (2021–2030), entered into force in April 2018 (European Parliament and Council 2018).

4.3 EU ETS Fit for Life? Phase 4 Reform

With the adoption of the MSR Decision and the revised EU ETS Directive for Phase 4, the European institutions have tried to tackle a number of issues. The

[10] To clarify what qualitative and quantitative restrictions were imposed, see: European Commission (2015).

two reforms aimed at addressing the historical surplus of EUAs, and making the EU ETS supply more responsive to changes in demand and able to deal with future oversupply.

Furthermore, the legislators sought to create a stronger link between free allocation and the actual production levels, as well as increasing the funds available for innovation and modernisation, to be financed through ETS-revenues (Marcu et al. 2018).

Some of the more significant measures taken are:

- increase of the LRF from 1.74 to 2.2% starting from 2021;
- alignment of the cap with a 43% GHG emissions reduction target by 2030, compared to a 2005 baseline;
- creation of a Market Stability Reserve to tackle the historical surplus of allowances on the market, while improving the EU ETS responsiveness to future shocks;
- phase-out of free allocation for those sectors not deemed at high risk of carbon leakage by the end of Phase 4;
- establishment of innovation and modernisation funds to support the energy transition;
- more flexible rules to better align the level of free allocation with actual production levels;
- benchmarks for free allocation to be updated twice during Phase 4, to avoid windfall profits and better reflect technological innovations (European Commission 2019g).

The introduction of a quantity management mechanism as the MSR deserves particular attention. Indeed, the MSR acts on the EU ETS through adjustments to the supply of auctioned allowances, increasing or reducing the supply of EUAs according to some pre-set threshold that reflect the total number of allowances in circulation. Starting from 2023, the MSR will invalidate the allowances held in the reserve at that point in time, if these exceed the previous year's auction volume.

This instrument is designed to eliminate the built-up surplus while improving the overall resilience of the EU ETS to market future imbalances. The key rationale for having an MSR is to prevent the formation of new surpluses on the market, as happened during the EU ETS Phase 2 and Phase 3 (ICAP 2019a).

4.4 Lessons Learnt and Way Forward

All in all, the recent reforms of the EU ETS framework follow almost 15 years of changes and adjustments to the European carbon market. The EU institutions have repeatedly tried to improve the functioning of the system, seeking solutions to the different challenges that emerged throughout the years.

Looking forward, it remains to be seen whether the Phase 4 reform and the MSR will bear the expected fruits, and if the EU ETS will continue representing a core element in the EU's environmental strategy towards 2030 and beyond.

Since the start of the EU ETS in 2005, many important lessons were learnt, especially with regard to the allocation of allowances and the risk of overcompensation. To fix these issues, different legislative reforms put increasing emphasis on auctioning as opposed to free allocation, with the latter moving from grandfathering to the use of benchmarks in order to reflect more closely the carbon intensity levels and efficiency of covered installations, while also rewarding early action, flexibility, and level of production.

Another key innovation was the recent introduction of the MSR, which should be welcomed as a positive attempt to shield the EU ETS from the effects of old and new sources of imbalance. The increasing attention to mobilising funding for innovation and modernisation is also an interesting development that could further encourage decarbonisation.

To evaluate the experience of the European carbon market, it is worth noting that the EU ETS was never a standalone policy: it operates in a highly interconnected environment and is affected by climate change and other policies at the global, EU, and EU MS levels (Marcu et al. 2019). Its performance should be analysed against this background, and not compared to an ideal world.

At the time when the EU ETS was launched, there was a widespread perception that other countries and jurisdictions would follow suit, and that there would be opportunities for international collaboration under the umbrella of the Kyoto Protocol. The EU ETS was crafted with the ambition of becoming a reference model for emissions trading, in a world where carbon trading would be broadly adopted as a leading solution to tackle climate change.

These expectations were partially frustrated. Internationally, the failure of adopting a new global climate agreement until the Paris Agreement in 2015 led to competitiveness concerns taking a central stage in the discussion on climate policy for many years (Stenegren 2018). At the same time, other ETSs emerged in different jurisdictions, proposing new models and different approaches to the trading of carbon.

At the EU-level, the domestic problems of the EU ETS supply-demand balance limited its function as the main driver of the EU's decarbonisation strategy. The weakened price signal in the market contributed to the EU ETS not playing a central role in the decarbonisation of ETS-sectors, as initially intended. The recent price increase of 2018–2019 seems to indicate a reversal of this trend, but it remains unclear whether this reflects a structural change or just a temporary trend.

With the urgency of tackling climate change becoming increasingly evident, the EU ETS will have to face new challenges in the future. In its 1.5 C report, the Intergovernmental Panel on Climate Change (IPCC) warned the international community about the impacts of climate change, urging countries to cut their emissions to net-zero by 2050 (IPCC 2018).

The EU followed this call and decided to embrace a carbon neutrality goal for 2050. The EU ETS 2030 target was significantly revised to reach an overall reduction of 55%. This translated into the adoption of an emissions cap of 61% by 2030 in the ETS sector, and a LRF of 4.2%. Expanding the scope of the system to include the transport and buildings sectors is envisaged (von der Leyen 2019).

5 Other Major ETS Markets and Potential for Interconnection

5.1 Emissions Trading Systems: Examples from Around the World

The EU ETS is not the only living example of an emissions trading system. Other ETSs have been in place for a number of years, each with specific design characteristics and different stories of successes and setbacks. This section will present an overview of the most significant experiences with the trading of carbon outside Europe, paving the way for a discussion on the potential for interconnecting different ETSs, also taking into account Article 6 of the Paris Agreement.

National or sub-national ETSs are already operating in many parts of the world. The focus will be on the examples of Canada, China, South Korea, and the United States.

At the time of writing, three out of four of these ETSs only apply at the sub-national level:

- Québec and Nova Scotia in Canada;
- Chinese city and provincial pilot projects in Beijing, Chongqing, Fujian, Guangdong, Hubei, Shanghai, Shenzhen, and Tianjin;
- In the United States the Regional Greenhouse Gas Initiative (RGGI)[11] and California.[12]

The different systems will be analysed using as a metric the main design options of an ETS, as outlined in Sect. 3 of this chapter.

5.1.1 Western Climate Initiative (WCI): California and Québec

Since 2014, the ETSs of California and Québec are linked under the umbrella of the Western Climate Initiative (WCI). We will therefore analyse them together.

[11] The states involved are Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Rhode Island, and Vermont. Since 2018, Massachusetts also complements the RGGI to help ensure it achieves its mandatory mitigation targets.

[12] Other ETSs in force or scheduled for implementation: Switzerland, New Zealand, Mexico, Colombia, Kazakhstan, Ukraine, Japan (city-level systems in Tokyo and Saitama). Jurisdictions considering the implementation of an ETS: Brazil, Chile, Russia, Turkey, Taiwan, Vietnam, Thailand, Indonesia, Japan. To clarify further, see: ICAP (2019c), 'Emissions Trading Worldwide—Status Report 2019'.

Compared to other existing ETSs, the WCI has the largest coverage, accounting for approximately 80% of the two jurisdictions' GHG emissions.[13]

California's cap-and-trade programme was launched in 2012, with compliance obligations starting from 2013. The programme has a mixed point of regulation for emissions above a certain threshold, which applies both downstream to electric power plants and industrial installations, as well as upstream by targeting fuel distributors. The inclusion of fuels distributors allows to incorporate more sectors into the system, such as transportation and building energy use (Hausfather 2017).

The allocation of allowances takes place through both auctioning and free allocation, roughly in equal shares (ICAP 2019c). Free allocation is based on benchmarking, in an attempt to reward efficient facilities while protecting them from the risks of carbon leakage. However, investor-owned utilities receiving free allocation are required to use the value of allowances to benefit ratepayers and achieve GHG emissions reductions (Center for Climate and Energy Solutions 2019a).

The overall GHG reduction target is set at 40% by 2030, from 1990 levels. To meet this target, the yearly cap declines at 4.1% between 2021 and 2030. In terms of flexibility mechanisms, banking is allowed under certain rules, borrowing is prohibited, and domestic offsets for projects in the US territory are allowed up to 8% of a facility's compliance obligation (California Environmental Protection Agency 2015).

Covered entities must report emissions annually, which are then submitted to an independent third-party evaluation. To avoid excessive price volatility, the Californian ETS has both a price floor and price ceiling. This means that the price of allowances cannot go below a pre-determined minimum. The state will sell an unlimited amount of emission permits if the price of carbon would reach the ceiling (Hausfather 2017).

Being linked to the Californian ETS, Québec's system has very similar design characteristics. Both systems cover the same greenhouse gases and sectors, set the same emissions thresholds and point of regulation, and have similar environmental targets (Kroft and Drance 2015). The allocation methods are also comparable in terms of the use of benchmarks and combination of free allocation and auctioning. However, Québec auctions a higher share of allowances than California, up to 70% in 2017 (ICAP 2019c). Price control mechanisms are identical under the WCI (Purdon et al. 2014).

5.1.2 Regional Greenhouse Gas Initiative (RGGI)

The RGGI was the first mandatory cap-and-trade programme in the United States. The system was first established in 2005, and became operational in 2008. It aims at limiting emissions from the power sector, and requires fossil

[13] Ontario has also been part of the WCI but has decided to termite its commitment in 2018. On the contrary, Nova Scotia entered the WCI since the beginning of 2019. The case of Nova Scotia will not be analysed, given that the ETS has been in place for only a few months.

fuel power plants with capacity greater than 25 megawatts to surrender an allowance for each ton of CO_2 emitted annually (Center for Climate and Energy Solutions 2019b).

Compared to the WCI, it only covers CO_2, and applies only to the electricity sector, with a downstream point of regulation. This translates in a coverage of less than 20% of the overall GHG emissions in those jurisdictions.

RGGI allocates 100% of the allowances via full auctioning, making it in theory the most economically efficient system (Narassimhan et al. 2018). A low marginal abatement cost, however, might also show that the system is tackling emissions from a sector where abatement opportunities exist and are available, not addressing other sectors where cutting emissions might be more challenging.

The system was reviewed in 2017. The new rules establish that between 2021 and 2030 the cap will reduce by 30% from a 2020 base year (ICAP 2019c).

Similarly to the WCI, the RGGI has price management mechanisms in place. The system has a price floor as well as a cost containment reserve, which releases allowances to the market when certain price triggers are reached.

Finally, RGGI states have been investing their auction revenues in programmes which benefit consumers: energy efficiency, renewable energy, direct energy bill assistance, and other greenhouse gas reduction programmes (ICAP 2019d). This has helped increasing the social acceptability of the system.

5.1.3 Korean Emissions Trading System (KETS)

The KETS has been operational since 2015 and represents East Asia's first nationwide mandatory ETS. The ETS covers almost 600 of the country's largest emitters, accounting for approximately 68% of the national GHG emissions (ICAP 2019c). Its coverage and scope include several gases and sectors, and emissions are regulated downstream.

One notable characteristic of the KETS compared to other ETSs is that it includes not only the power generation sector, but also indirect emissions from electricity use. The reason for this is that the electricity price in Korea is rigid and controlled by the government, so it would not automatically provide carbon cost pass-through. Therefore, the efficient use of electricity is encouraged by including electricity consumption in the ETS, thereby providing a direct price signal to consumers (Asian Development Bank 2018a).

The KETS cap reflects an environmental target of cutting emissions by 37% by 2030 compared to a business as usual (BAU) scenario, that is, a 22% reduction below 2012 GHG levels (NDC) (ICAP 2019e). This cap was set through a bottom-up approach. This has attracted the criticism of some analysts, considering the heavy reliance on reported data from manufacturers (ICAP 2019c).

To meet the KETS target, up to 38 million international credits may be used for compliance, including international credits through the KETS, as well as alternative options as land use, land use change and forestry (LULUCF) and other international credits (i.e. under Article 6 under the Paris Agreement) (ICAP 2019c). The positive approach of Korea towards the use of international credits can be explained by one issue which has represented a problem for the KETS since the beginning of its operations: a general lack of market liquidity.

Indeed, being the Korean market fairly limited in terms of market participants, it has suffered from the unwillingness of its participants to sell unused allowances, as they preferred to bank for future compliance periods (Lee and Yu 2017). Connecting the KETS with other ETSs through linking or expanding the acceptance of international credits is perceived as a way to increase liquidity and increase the trading volumes.

Opposite to the example of the RGGI, allocation under the KETS takes place through free allocation for 97% of the total allowances put on the market. This has helped the initial public backing for the system, yet to the detriment of price discovery due to low liquidity in the secondary market, and no revenue generation for the government. Similarly to the RGGI and the WCI, the KETS has market stability provisions that adjust the supply of allowances according to some pre-set price thresholds.

5.1.4 China

The prospect for China to introduce a nation-wide ETS has long been discussed, attracting a lot of attention given China's role as the world's largest GHG emitter. In 2013, China initiated pilot ETSs in seven regions, aiming at the future creation of a national system. In 2016, an eighth pilot ETS was launched in the province of Fujian (Zhang et al. 2019).

Since 2017, the Chinese national ETS was politically launched, and the first trading operations started in July 2021. As soon as trading began, the Chinese ETSs became the world's largest carbon market, almost doubling the size of the EU ETS, although it only includes the power sector initially, but the scope should be gradually expanded to other industries (ICAP 2019f).

The analysis of the pilot projects can give us some guidance on how a Chinese ETS might look like, and what design characteristics would the system have. For all of them, the decision was to set absolute-based caps (with the exception of Shenzhen, which adopted both an absolute and intensity cap), with different degrees of stringency and linear reduction factors. The programmes account for ca. 30% of China's GDP and 17% of the country's CO_2 emissions (ICAP 2019c).

Most pilots focus on CO_2 emissions, whereas they all include both power installations and some industrial sectors. The vast majority of the pilots adopt free allocation as the main allocation method, though several are also opening up to auctioning allowances. In terms of the way allowances are freely allocated, grandfathering seems to emerge as the default option, with only the Shenzhen ETS using benchmarking for allocation (ICAP 2019c).

So far, carbon prices have been relatively low, affected primarily by generous levels of allocation. Furthermore, the credibility of the system has been put into question due to limited market transparency and access to data. On the other hand, the pilot programmes are having the positive implication of legitimising the concept of carbon trading in China, training local governments officials and making MRV standards mainstream. This is preparing the way for the potential adoption of a national system (Timperley 2018).

5.2 Carbon Trading and Potential for International Cooperation Mechanism

Section 3.4 highlighted that ETSs can increase flexibility and increase economic efficiency, effectively lowering costs for abatement by:

(a) accepting domestic offsets for compliance;
(b) accepting international offsets for compliance;
(c) linking different ETSs—that is, participants in one system can use a compliance instrument issued by another ETS for domestic compliance.

The idea of interconnecting different systems through the international trading of carbon has been at the centre of the international debate on climate change since the adoption of the Kyoto Protocol.

Article 17 of the Kyoto Protocol (KP) states:

> *The Conference of the Parties shall define the relevant principles, modalities, rules and guidelines, in particular for verification, reporting and accountability for emissions trading. The Parties included in Annex B may participate in emissions trading for the purposes of fulfilling their commitments under Article 3. Any such trading shall be supplemental to domestic actions for the purpose of meeting quantified emission limitation and reduction commitments under that Article.*

Art. 17 has represented the umbrella for the creation of an international carbon market, including the trade of international offsets, and linking of different ETSs.

When discussing the potential for linking, it was mentioned that creating a bilateral link has the main benefit of reducing the total cost of achieving the combined emissions target of the linked ETSs (Partnership for Market Readiness 2014). Additionally, linking could prove particularly beneficial for small jurisdictions, where setting up an independent ETS could lead to high technical and compliance costs, with likely problems of market liquidity.[14]

At the same time, linking can also cause problems, as it changes the distribution of costs in each system. This leads to price convergence, benefitting buyers with the higher pre-link price and sellers with the lower pre-link price (ICAP 2019c).

To limit these issues, the linking of different systems should be carefully designed, and both jurisdictions will need to abide by similar, if not identical,

[14] This was one of the main drivers behind the agreement between the EU and Switzerland to link their respective ETSs. To clarify further, see: ICAP (2019g).

standards. Indeed, the experience of the California-Québec linking suggests that linking is easier if certain design features are aligned. The same can be said also of the EU ETS, where the distribution of allowances and operation of the registry have been gradually centralised, following initial years when the EU Member States were in charge of these operations even if this was one ETS.

Overall, different jurisdictions will be open to the possibility of linking for as long as they have reasonable certainty that the system they are linking to has similar design and objectives. Besides the technical considerations, linking also relies on the political will to cooperate with a foreign jurisdiction on climate change policy, as it implies partially losing control over the ETS. Domestic support for emissions trading and linking is therefore a crucial pre-requisite to the decision to linking different ETSs (Beuermann et al. 2017).

The other possibility for the international trading of carbon is via international carbon offsets. According to the United Nations, *offsetting is a climate action that enables individuals and organizations to compensate for the emissions they cannot avoid, by supporting worthy projects that reduce emissions somewhere else* (UN 2019). Historically, in the context of the UNFCCC, offsetting could be done by industrialised nations, mentioned in Annex I of the KP, as they were the only ones who had committed to caps under Kyoto. The KP created three main categories of offset credits: emission reduction unit (ERU) generated by joint implementation (JI) projects; certified emission reduction (CER) generated from clean development mechanism (CDM) project activities; and removal unit (RMU) generated by LULUCF activities.

Offset credits can also be produced outside the UNFCCC. These include voluntary offset programmes (e.g. Verified Carbon Standard), national offset programmes (e.g. Australia's Carbon Farming Initiative), bilateral offset mechanisms (e.g. Japans' Joint Crediting Mechanism), and regional offset programmes (e.g. Climate Action Reserve offsets allowed under California's cap and trade scheme) (Carbon Market Watch 2013).

Primarily, international offsets must ensure environmental integrity in order to be effective. To this end, it has long been recognised that offset credits should be 'real, permanent, additional and verified'. Indeed, environmental integrity of offset credits is guaranteed for as long as:

- emissions reductions have actually occurred, and are not merely artefacts or incomplete/inaccurate accounting—'real';
- reductions are permanently removed from the atmosphere and/or are backed by replacement mechanisms if they are re-emitted to the atmosphere—'permanent';
- the emissions reduction that underpin the credit would not have occurred in the absence of the activity that generates the credit—'additional';
- reductions result from projects that can accurately be monitored and verified—'verified' (Gero 2009).

Failure on one or more of these dimensions would risk undermining the credibility and effectiveness of an offset. Notably, credits under the KP were sometimes criticised by campaigners who claimed that credits under the KP suffered from shortfalls in terms of environmental integrity, accounting, and contribution to sustainable development.

A new system for the international trade of carbon is now being developed as part of the negotiations for the Rulebook of Article 6 of the Paris Agreement (PA), which were successfully concluded at COP 26 in Glasgow (Marcu 2021).

Article 6 provides a framework for general cooperation in the implementation of the Paris Agreement and the nationally determined contributions (NDCs), including provisions to create a framework that will enable the creation of an international carbon market.

One key difference between the PA and the KP is that under the PA all countries, both developing and developed countries, have to submit NDCs as part of the joint effort to reach the overall objective of the Agreement. This implies that potentially all countries might be interested in the international trading of carbon, as both buyers and sellers.

As a consequence, before engaging in the international trading of carbon under Art. 6, countries will need to agree on how to ensure environmental integrity, how credits might or might not account towards the achievement of a country's domestic NDC, and how double counting can be avoided on the basis of corresponding adjustments (Asian Development Bank 2018b).

More broadly, the open question for the future is whether carbon markets will play a central role in the fight to climate change, and if there is room for the emergence of a global carbon price. Developments of new ETSs and the outcome of the negotiations on Article 6 of the PA will reflect the attitude of the international community towards the use of market-based approaches to climate policy.

If anything, the emergence of a global carbon price would have the undisputable benefit of alleviating carbon leakage concerns.

The ongoing development of new ETSs in high-emitting countries like China could be seen as a step in the right direction, as having more jurisdictions effectively pricing carbon decreases the potential for unfair competition. However, it remains to be seen if the new emerging ETSs will apply high standards of environmental integrity, triggering ambitious decarbonisation strategies in their respective jurisdictions.

As for international cooperation mechanisms under Art.6 of the PA, this could represent a steppingstone for a resurgence of the trading of international carbon credits, potentially helping the international harmonisation of carbon pricing (Asia Society Policy Institute 2018).

Finding an agreement on the operationalisation of Art. 6 is therefore extremely important, provided that such agreement shapes a system that works for the environment and helps countries achieving the objectives of the PA. All in all, while markets do adapt to many circumstances, they will be attractive only if they have clear objectives, have governance that ensure predictability, are liquid and transparent, and participants are re-assured that they have clear ownership of the assets (Marcu 2019).

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