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Domestic Emissions Trading Systems

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Introduction

The EU ETS is the centrepiece in Europe's climate policy architecture and the largest cap-and-trade system in the world. After having failed to introduce a union-wide carbon tax during the 1990s, the European Commission presented a green paper in 2000 that proposed the use of emissions trading. Launched in January 2005, the EU ETS established a uniform carbon price for specific heavy-industry activities in all EU member states. It covers CO_2 emissions from over 10,000 installations, including power and heat generators, oil refineries and factories for ferrous metals, cement, lime, glass and ceramic materials, and pulp and paper. Together, the covered sources account for roughly 40 per cent of total EU GHG emissions.

This chapter reviews the characteristics and experience of domestic emissions trading as an instrument of climate policy. The rationale of emissions trading and its general principles are briefly summarized first. Then we highlight the basic design elements of cap-and trade, and then review the main lessons from the EU ETS. Finally, we summarize the conclusions.

Rationale of emissions trading

At the very heart of emissions trading lies the notion that if 'factors of production are thought of as rights, it becomes easier to understand that the right to do

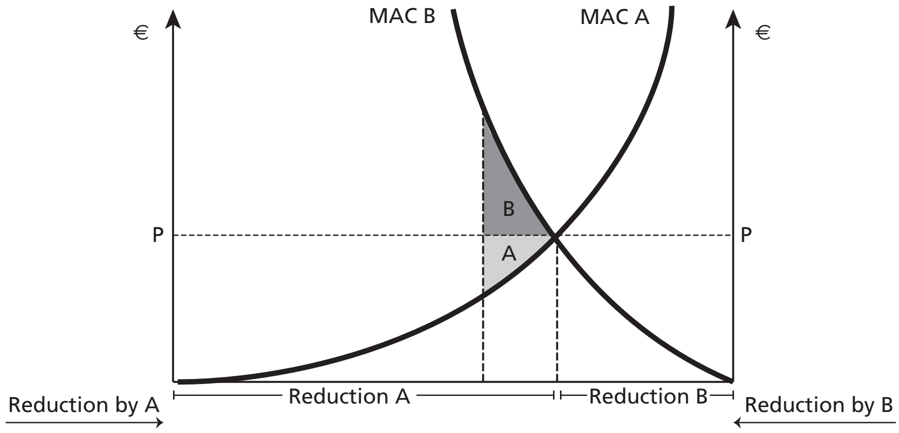

Figure 8.1 *Efficiency-enhancing effect of emissions trading*

something which has a harmful effect … is also a factor of production' (Coase, 1960). The cost of exercising such a pollution right should be equivalent to the loss that is suffered elsewhere in consequence of this action. Making these rights explicit and transferable enables the market to value and trade them. As a result, emission rights can flow to their highest value use and reductions can occur wherever they are cheapest. Figure 8.1 below illustrates this effect. Imagine polluter A with a MAC curve A, and polluter B with a relatively steeper MAC curve B. Given a certain emission limit (left dotted line) and without emissions trading, both polluters would have to reduce emissions until they comply with their individual abatement obligation. Polluter A would face relatively low cost, polluter B relatively high cost for a given quantity. With emissions trading, A will want to increase and B decrease abatement until marginal abatement costs converge at price P. The efficiency gain of A is illustrated by area a, which comprises the benefit from increasing abatement and selling freed up allowances to B. The efficiency gain of B is illustrated by area b, which represents the amount of saved abatement cost net of the money paid to A.

After the groundbreaking work of Coase in the 1960s, scholars quickly began to point out the instrument's usefulness for air pollution control. Tradable emission permits have been successfully used in the US since 1995 to tackle the problem of acid rain by reducing sulphur dioxide and nitrous oxide emissions.

Two broader categories of domestic ETS can be differentiated: systems that limit emissions at relative or at absolute levels (Reinaud and Philibert, 2007).[1] Relative targets can either apply to country or sector/firm level. It is also possible to combine relative targets for some sectors with an absolute target at country level. Relative targets can be based on emission intensity per unit of output or other indexes and have been proposed as a means to address cost concerns as they better encapsulate the notion of decoupling economic growth from emissions (Marschinski and Lecocq, 2006). Ellerman and Sue Wing

(2003) contend that intensity targets can reduce the uncertainties associated with abatement costs under uncertain economic conditions. Pizer (2005) finds that intensity targets may be more appropriate if the short-term objective is to slow, rather than halt, emissions growth.

Systems based on absolute targets cap emissions from covered entities and allocate emission allowances to participants that can then trade them among each other. Under cap-and-trade, compliance is established by comparing total actual emissions with the amount of allowances a particular participant holds. Dudek and Golub (2003) argue that absolute caps have more certain environmental results and lower transaction costs for emissions trading, thereby creating stronger incentives for technological change. IPCC (2007a) concludes that absolute targets are more effective than intensity targets if the aim is to meet certain emission reductions, but they may be less effective at addressing costs when the economic outlook is uncertain. The majority of currently existing, announced and proposed schemes apply cap-and-trade and the remainder of this chapter therefore focuses on this type of emissions trading.

Elementary design features of cap-and-trade

In theory, there exists a wide range of different cap-and-trade system designs. Table 8.1 in the appendix to this chapter summarizes the major characteristics of the EU ETS, the Regional Greenhosue Gas Initiative (RGGI), and proposals for systems in California, Australia and New Zealand. While they differ widely in their specific rules, they all build on a limited number of design elements that enable policy-makers to strike a balance between a programme's environmental effectiveness, cost effectiveness, distributional impact and institutional feasibility.

Cap and coverage

The coverage identifies the scope of an ETS in terms of emission sources and GHGs. Emissions sources are usually covered at sector level, where minimum thresholds are defined to exclude very small sources. This can avoid that transaction costs of including additional sources exceed the environmental benefits.[2] The cap defines the maximum level of GHGs these sources may emit during a certain period of time. Several studies have found that full coverage of emissions sources within an ETS is superior to partial coverage because it ensures that marginal costs are equalized across the entire economy. Böhringer and Löschel (2005) find that in the EU significant cost savings would result from economy-wide coverage compared to exclusion of certain sectors. Given the substantial abatement options in different economic sectors such as buildings and transportation (Enkvist et al, 2007), it is important to make these options accessible for the market by covering all relevant sectors by the cap. This can be done upstream in order to minimize transaction cost. The EU ETS, however, excludes buildings and transport from coverage, leaving their abatement potential inaccessible for the trading sector.

Complementary measures such as insulation and fuel efficiency standards should be employed in order to exploit the abatement potential of non-covered sectors. In fact, the presence of negative abatement costs indicates that market failures exist that impede the exploitation of options that would be beneficial even in absence of climate policy. However, it needs to be taken into account that as long as such activities are not subject to a binding constraint on overall emissions, for example through a cap, rebound effects will occur. As higher efficiency reduces the costs of an activity (for example transport or heating), this will lead to an expansion of the activity level. Thus, emission reductions may not be as large as suggested by initial assessments of reduction potentials from efficiency measures that do not take into account rebound effects. Moreover, even if a certain sector previously excluded becomes covered, complementary policies continue to be necessary in order to address the pre-existing market failures indicated by negative abatement costs. Both an absolute cap on emissions and complementary instruments are required to reduce emissions in a cost-efficient manner.

Broad coverage can also widen the potential for carbon market liquidity and efficiency (Baron and Bygrave, 2002). Introducing more sources and sectors to a market can also reduce the impact of economic shocks in any one sector on the trading system as a whole. However, this comes at the expense of partly transferring shocks into other sectors via the permit market. In addition, the larger and more liquid a market, the harder it is for large players to affect the overall price level. However, sectors for which emissions cannot be quantified and monitored with certainty (for example N_2O and CH_4 emissions from agriculture) should be managed with alternative measures and remain excluded from the cap until sound methodologies of measurement and monitoring have been developed. Otherwise, the inclusion of unverifiable emissions could undermine the cap's environmental integrity (Herold, 2008).

Another key aspect in designing ETS is identifying the appropriate incidence of regulation. The point of obligation defines the liable party for surrendering emissions permits. In an upstream design, the point of obligation can for example lie at the level of importers and producers of fossil energy.[3] Small energy-related sources, for example in the transport sector, can be effectively covered by moving the point of obligation upstream. Importers and producers of fossil resources would then pass on the price signal to resource users, who will adapt their economic activities according to this change of input prices. In a downstream design the point of obligation can be the emitting installation or the final consumer of fossil energy. A hybrid system combines both approaches, where some sectors are included upstream and others downstream (Hargrave, 1998).

While CO_2 related to the combustion of fossil fuels accounts for the bulk of man-made emissions, several other sources and compounds also contribute significantly to global warming. Currently, CO_2 release due to deforestation, mostly in tropical countries, accounts for almost 20 per cent of the emissions (IPCC, 2007b). In terms of CO_2 equivalents, emissions of methane and nitrous

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oxide from agriculture make up 14 per cent of the global total. Agricultural emissions have risen continuously and are particularly difficult to reduce. Other significant sources are nitrous oxide and fluorinated gases from industrial processes. Many of these emissions can be reduced at low costs. A major concern often voiced is that including low-cost abatement opportunities and offset credits (see below) will, in the short term, crowd out other mitigation activities and thus delay the structural energy sector transformation that is necessary for the long-term achievability of ambitious climate stabilization targets. This problem is mitigated by setting appropriately tight short- and medium-term reduction targets in order to ensure sufficient incentives for low-carbon energy investments and inter-temporal cost optimality, while taking advantage of low-cost emission reduction opportunities.

Generally, governments need to be as clear as possible about the future development of cap and coverage, as they are the major determinants for emission prices. For making long-term investments with confidence, market participants require predictability. Clear signals of support for stringent future caps from governments will enhance predictability. This requires in particular the definition of a long-term reduction trajectory. After all, GHGs are stock-pollutants. Their climate effect depends primarily on the long-term emission budget rather than reduction targets for certain points in time.

Importing offset credits

ETS coverage can also be extended by allowing for the import of offset credits. Emission reductions in sectors or countries not covered by the ETS can be certified and redeemed to offset emissions under the cap. The CDM, for example, is the largest offset credit programme in the world with a market value of $13 billion in 2007 (World Bank, 2008). Allowing the import of offsets will lower compliance costs if the marginal costs of creating credits are lower than the allowance price in the cap-and-trade system. This might be sensible for sectors in which emission levels and reductions are difficult to measure for all sources, such as forestry, agriculture and waste. Partially covering these sectors in the ETS might lead to intra-sectoral distortions, whereas allowing them to generate project-based credits can provide incentives for mitigation without generating distortions (Garnaut, 2008). The planned Australian ETS, for instance, allows for domestic forestry offsets in order to provide incentives for this sector before it becomes eventually covered under the scheme. The EU is also planning a new domestic offset mechanism from 2013 onwards (EC, 2008), and the RGGI system also includes a domestic offset credit system (RGGI, 2007).

There are, however, risks connected to the import of credits. First, the excessive reliance on offsets to meet abatement obligations may jeopardize a programme's environmental effectiveness. This concern arises when the additionality of offset credits cannot be fully guaranteed.[4] Offsets are rewards for reductions in emissions measured against an assumed baseline: 'what emissions would have occurred in the absence of the credit mechanism?'.

Determining a counterfactual baseline can be arbitrary to some extent, resulting in the risk that schemes do not promote genuine abatement. Second, the import of relatively cheap abatement options via offset credits drives down allowances prices under the cap, which in turn reduces the financial incentive for investing in domestic emission reductions. This can be a problem if a certain level of domestic abatement and certain incentives for investments are an ETS policy objective. From an environmental point of view, however, it does not matter where GHG emissions are reduced internationally as they equally contribute to global warming. To address these previous concerns, an ETS can limit the amount of credits that can be used. The EU, for example, limited the import of Certified Emission Reductions generated under the CDM into the EU ETS to 13.4 per cent. Limiting the import of credits does, however, not address the problem of additionality (except when limiting the import quota to zero), and can only contain the scale of the problem. Additionality concerns are best addressed by applying rigorous additionality procedures, and/or reforming the credit scheme accordingly. Discounting offset credits is another option to address uncertainty over project additionality.

Allowance allocation

After the cap has been set, the ETS operator issues allowances for units of emissions (for example metric tonnes CO_2[5]) where the total number of permits equals the cap. This step is a contended issue in virtually every cap-and-trade programme as it critically determines the policy's distributional impact. Allowances can either be sold, usually through auction, or allocated for free. The key question is, who receives the scarcity rent created by capping emissions?

In principle, there are two options for free allocation: grandfathering and benchmarking. Grandfathering allowances means that permits are distributed in proportion to past emissions, measured for one or several years. Grandfathering can either be a one-off allocation to existing sources or be regularly updated with new emissions data. Benchmarking constitutes a method where allowances are distributed on the basis of an average or expected performance benchmark. For instance, it can provide fixed allocation quantities based on expected output. Under free allocation, emitters receive the full value of allowances for free and pass on the opportunity costs of allowances to consumers wherever possible. The scarcity rent of allowances is thereby appropriated by the private sector, generating windfall profits for emitters and higher output prices for consumers.

There are a number of reasons why free allocation may distort incentives for reducing emissions (Hepburn et al, 2006). First, the expectation that the baseline year upon which allocations are based will be updated may encourage sources to invest in dirty technology or refrain from investing in clean technology in order to increase or maintain emissions levels and thus receive more free allowances in future. One needs to be cautious to avoid this kind of perverse incentive, for example by choosing a baseline year in the past or by combining baselines with benchmarks. Second, free allocation may present a barrier to market entry. If existing polluters receive allowances for free but new entrants are obliged to buy them, free allocation directly disincentivizes market entry, reducing competition. At the same time, it may present a barrier to market exit. The requirement that an installation must be kept open in order to receive free allowances may prevent the closure of inefficient plants, freezing emissions at higher levels than otherwise necessary. Third, grandfathering leads to increased lobbying activities because emission allowances, which have a monetary value, are given out for free. Lobbying of powerful producer groups may put governments under considerable pressure – a time-consuming and costly procedure.

Auctioning offers several advantages over free allocation. First, it puts upfront costs on polluters. This will tend to enhance management's awareness of carbon cost, leading to more efficient decisions. Second, auctioning enables governments to use revenues to address equity issues through reductions in taxes or other distributions to low-income households. Governments can also use auctioning revenues to invest in the development and deployment of cleaner technologies, or provide finance for other countries' efforts for climate change mitigation and adaptation. This is analogous to the 'double dividend' feature of carbon taxes. Goulder et al (1999) demonstrate that recycling revenues from auctioned allowances can have economy-wide efficiency benefits if they are used to reduce distortionary taxes. Dinan and Rogers (2002) show that free allocation of tradable permits may be regressive because it leads to income transfers towards higher-income groups (i.e. shareholders) at the expense of poorer households (i.e. consumers with high income shares of energy expenditure). Third, Tietenberg (2006) contends that auctioning provides stronger incentives for technological innovation. Under grandfathering some sources are buyers and others sellers. Sellers have an incentive to behave strategically and keep prices high by avoiding technological innovation. In auction markets, all emitters are buyers. Hence, all sources benefit from low-carbon technologies via decreased marginal abatement cost and permit prices.

Allocation methods for sources entering or leaving the scheme represent a contentious issue. New entrants and closure provisions define whether new entrants have to either buy/receive additional allowances from a new entrant's reserve or be obliged to buy permits on the market (Ahman et al, 2006; Ellerman, 2006). Free allocation would be more favourable to encourage new investments, as the obligation to buy allowances represents an additional barrier to market entry and might impede the renewal of facilities. However, if free allocation to new entrants is based on actual emission intensity, it provides a disincentive to install more efficient technology, contrary to the overall aim of carbon pricing. Grubb and Neuhoff (2006) therefore suggest basing free allocation for new entrants on the basis of technology-specific benchmarks.

Overall, auctioning can avoid many of the problems associated with free allocation and offers distinct advantages. But good design is necessary to avoid new inefficiencies. Small frequent auctions may be more effective in limiting

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the market power of large bidders (Hepburn et al, 2006). They may also encourage learning processes, help players to adjust bids and promote price stability. In contrast, one large auction at the beginning of each trading period may minimize administrative costs. However, it may also enable large polluters to buy the bulk of permits and then use them to extract oligopoly rents on the secondary market. The optimal design for auctions in cap-and-trade systems is subject of ongoing research (Neuhoff and Matthes, 2008).

Banking and borrowing

Economic theory states that the efficiency gains from emissions trading stem from equalizing marginal abatement costs among spatially distributed sources with different abatement cost curves. In addition, however, there is also a temporal dimension to equalizing marginal abatement costs. Due to economic cycles/shocks, resource price hikes, mild winters and so on, allowance prices can be low in one particular period of time and high in another. However, it would be efficient to reduce emissions when this is cheapest. Theoretically, if the cumulative emission budget remains constant and it does not matter in which period emissions are reduced, this will enhance the efficiency of meeting a predefined emissions target. In cap-and-trade systems, temporal flexibility is introduced by banking and borrowing provisions.

Banking means that instead of using an emissions permit at an early date, it is saved for later use because it is anticipated that its value in later periods will be higher than now. As a result, abatement options that are cheaper now than the net present value of a future allowance will be utilized, thus increasing the amount of early abatement relative to the case without banking. Allowance banking can help to equilibrate present-value prices across trading periods. Ellerman and Pontero (2005) analyse the use of banking in the US Acid Rain Program. They find that sources banked 30 per cent of permits in phase one (1995–1999), probably due to the expectation of tighter caps in later phases. As a result, emission reductions in phase one were twice as high as needed to meet the cap. Fell et al (2008) demonstrate that banking reduces price volatility and eliminates about 20 per cent of the cost difference between price and non-bankable quantity instruments. The degree of these reductions, however, depends on the persistence of price shocks. Due to these advantages, banking is a feature found in most ETS designs.

The advantages of borrowing, in contrast, are less unanimous. Borrowing means the ability to buy allowances from a future trading period during an earlier trading period. While in theory, borrowing can significantly enhance economic efficiency of achieving a cumulative cap, concern over the instruments' impact on policy consistency exists. As excessive borrowing undermines early abatement, it is likely to put upwards pressure on future compliance costs, unless substantial technological change occurs in the meantime. Regulators, faced with considerable political pressure from polluters who refrained from investing in low-carbon technologies early on, might be pressed

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to cancel out borrowed quotas by increasing future allocations.[6] Sticking to announced allocation budgets might thus become politically untenable. Assuming that borrowed quotas will not be repaid, the dominant strategy from the point of view of an emitter is to refrain from early abatement, create pollution lock-in and bet on loosened caps in the future. Excessive borrowing from future compliance periods could therefore undermine long-term policy integrity while also depressing short-term allowance prices and hence financial incentives to invest in low-carbon technologies for all other players. Nonetheless, some proposed emission trading programmes include borrowing as a means of cost containment.[7] The conditions under which borrowing can be in line with a stringent trading system and its environmental effectiveness remain subject to further research.[8]

Price ceiling and floor

As noted earlier, under uncertainty and in the near term, carbon taxes tend to be more economically efficient than emissions trading (Newell and Pizer, 2003). Yet, the introduction of carbon taxes has proven to be politically difficult, resulting in a greater emphasis on emissions trading. In order to combine the economic efficiency benefits of carbon taxes with the political feasibility advantages of emissions trading, some scholars have investigated how quantity controls can be reconciled to efficient price policies via the application of price bounds (Roberts and Spence, 1976; Newell et al, 2005).

A price ceiling, also called 'price cap' or 'safety valve', can help to limit cost uncertainty by defining a fixed price at which additional allowances are made available in excess of the cap (Pizer, 2002; McKibbon and Wilcoxen, 2002; Jacoby and Ellerman, 2004). A price floor, in contrast, defines a minimum charge on emissions. Price floors stabilize expectations for developers of low-carbon technologies as they are assured that abatement technologies below the price floor will be marketable. Most ETS designs do not include a price floor. One exception is the RGGI, which plans to define a minimum 'reserve price' for allowance auctions. Although this establishes a minimum price for allowances during auctions, it does not guarantee a minimum price during trading periods.

The underlying reasoning of price ceilings is that the atmospheric stock of emissions over the long term needs not necessarily be affected by a short-term deviation from the reduction path. There may therefore be less concern about short-term increases in CO_2 as long as the overall trajectory of CO_2 emissions is downward over an extended period (Newell and Pizer, 2004). Price ceilings can cap expected cost and hereby reduce economic uncertainty. However, the uncertainty would only be shifted from the cost side to the benefit side of reducing emissions. Philibert (2008) estimates that implementing price ceilings and floors in the global climate architecture can reduce mitigation cost by two-thirds (from $10,671 billion to $3456 billion for the 10.5Gt CO_2 scenario) without greatly increasing global warming (from 2.44°C median increase to 2.49°C by 2050).

Price ceilings, however, can put a cap's long-term environmental effectiveness at risk, especially where they are combined with banking. Suppose market participants expect the emissions cap to be tightened in the near future (for example due to new information on the severity of climate change). As a consequence, they also expect a higher price ceiling under the new cap. Market participants would want to buy as many allowances as possible at the current lower trigger price in order to bank them for later use. An unrestricted safety valve mechanism could thus put much stress on the stringency of long-term emission reduction targets (Stern, 2007). In addition, price ceilings generate additional administrative complexity, especially in regard to linking different trading systems. The lowest price ceiling among linked trading systems would set the international price ceiling (Flachsland et al, 2008). In order to safeguard the stringency of long-term caps, Murray et al (2008) propose to limit the quantity of additional allowances available in the safety valve. This 'allowance reserve' would stipulate both a ceiling price at which cost relief is provided and a maximum number of allowances available to provide this relief.

Monitoring and compliance

The provision of robust information is a necessary condition for ETS price stability (Stranland et al, 2002). Stringent monitoring, reporting and verification (MRV) procedures are required to reveal the basis of market supply and demand in a reliable manner, excluding later data adjustment. The price crash in the first trading period of the EU ETS (see below) resulted from the fact that prior to the implementation of the system no reliable emissions data were available. MRV rules ensure that a tonne of carbon emitted by one source is equal to a tonne of carbon reduced by another source. Hence, they are essential for securing the environmental credibility and the financial fungibility of emission permits. The installation of stringent MRV standards may pose a challenge, in particular for countries with weak legal traditions and institutions.[9] Tietenberg (2006) states that emissions trading systems introduce an incentive for cheating regarding the provision of emissions data. Under grandfathering, sources have an incentive to register more historic emissions than actually occurred. Depending on method of allocation, non-registered emissions do not need to be paid for (auctioning) or can be sold (free allocation). For ensuring compliance, excess emission penalties should be set at levels substantially higher than the prevailing allowance price (Swift, 2001). An additional incentive for compliance is provided by public access to data because third parties such as traders or non-governmental organizations (NGOs) can screen the available information for inconsistencies (Kruger et al, 2000).

Avoiding carbon leakage

In an ideal world, global emissions trading would establish a uniform carbon price for all countries and all sectors. But in reality, carbon pricing is only applied partially and unevenly, so that firms facing a cost on carbon might

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compete with firms who do not. Asymmetric carbon pricing gives rise to concerns over carbon leakage that occurs when domestic production is reduced as a result of carbon costs but (at least some of) the reduced domestic production gives rise to additional production and emissions elsewhere. Trade-exposed energy intensive industries might emigrate to non-carbon jurisdictions and/or carbon-constrained sectors might lose shares in the international market. Both outcomes reduce the environmental effectiveness of cap-and-trade. Concerns exist that both effects will also negatively impact competitiveness and employment of carbon-constrained sectors without necessarily benefiting the global environment.[10]

For assessing the risk of carbon leakage, one needs to distinguish between direct and indirect price effects generated by emissions trading (Grubb and Neuhoff, 2006). Direct costs are caused by emissions originating from the production process itself (which include energy and process emissions). Indirect costs are caused by, for example, higher electricity prices: electricity generators pass on opportunity costs of allowances to firms, which in turn will result in higher prices for consumption goods. The same indirect cost effect may apply to other intermediate inputs to production that may become more expensive due to pass-through of CO_2-related costs. The German government (UBA, 2008) assessed both effects for sectors that may potentially be exposed to distortions in competitiveness. The analysis is based on the concept of value at stake, which is defined as the sum of potential direct and indirect costs in relation to the gross value added (GVA) of a given industrial sector. The sectors with relevant value at stake contribute little more than 2 per cent to overall GDP. In other words, most economic sectors are not at risk of carbon leakage.

Grubb and Neuhoff (2006) argue that for most industries, the net value at stake due to adverse competitiveness impacts are lower than 1.5 per cent of sector value added, and that for EU industries that are affected the most (cement, and iron and steel) import penetration from outside the EU is rather low. This is in line with the findings of the World Bank (2007), which concludes that industrial competitiveness is not seriously impaired by domestic emissions reductions and that the remaining adverse effects can be offset by well-designed policy packages. Further, Reinaud (2008) finds limited risk of carbon leakage. Production factors other than carbon costs also influence investment and production decisions: cost and qualification of employees, exchange rates, infrastructure, existence of technology clusters and so on. With regard to most industrial sectors, carbon cost influences investment decisions only marginally.

For the limited number of sectors that are prone to carbon leakage, one countermeasure is free allocation of emission allowances. However, for those sectors for which the potential value at stake depends mostly on the increase of electricity prices (indirect CO_2 costs), a change in the allocation method would not prevent a potential production cost increase (UBA, 2008). For those sectors with high direct CO_2 costs, it should be noted that the maximum value at stake is based on the assumption of full auctioning of emissions allowances. Direct costs could be lower if part or all of the allowances were allocated for free, depending on whether, and to what extent, the concept of opportunity costs is applied. Without a functioning international carbon market, partial or totally free allocation for a limited number of industries prone to carbon leakage may be necessary to maintain a level playing field among international competitors.

Another measure that features in the discussion surrounding carbon leakage is trade sanctions, i.e. imposing a tariff or border tax on each imported good that is directly proportional to the carbon emitted during its production process. However, border tax adjustments hardly seem feasible in practice, as they would require a complete life-cycle assessment for each single product that is imported. Otherwise, it seems highly likely that imposing uniform carbon tariffs on all goods from all countries that oppose climate protection could be seen as 'arbitrary' and 'discriminatory' under WTO rules. To circumvent this last problem, Ismer and Neuhoff (2007) propose that border tax adjustment should focus on primary materials incorporated in imported products and, employing detailed data on production processes, assume that these products have been produced using the best available technology (BAT). However, it is not entirely clear how this proposal could be implemented in practice, what the implications for WTO law are, and how effective in reducing carbon leakage it could turn out to be.

Key lessons from the EU ETS[11]

The EU ETS runs in phases. Phase I (2005–2007) was planned as a test phase during which experience with the programme could be developed and banking of allowances was effectively not allowed. Phase II (2008–2012) coincides with the five-year Kyoto commitment period. In line with the EU-wide Kyoto target, each member state has its own national emissions target as defined by the EU burden-sharing agreement. Each member state is required to develop a National Allocation Plan (NAP) that addresses the national target in two steps. First, it splits the national reduction burden between ETS and non-ETS sectors, hereby setting the trading sector's cap. Second, it specifies how the allowances will be allocated to the sources in the trading sector.

Setting the cap

The EU-wide caps of Phases I and II where determined in a decentralized negotiation process between the EU Commission and member state governments. The aggregation of national caps as specified in NAPs resulted in the EU-wide cap. The Commission had the powers to review NAPs and reduced 15 NAPs in Phase I by together 290 million tonnes (Mt) annually. Although this cut aimed at enforcing scarcity in the system, Phase I caps were still too lax. NAPs were based on projected BAU emissions, many of which were overestimated due to the influence of firm lobbying and information asymmetries between companies and regulators. Covered sources in Phase I received emission permits for 2080Mt CO_2, while actually emitting 2020Mt CO_2 annually on average (EEA, 2007). Hence, the number of allocated emission allowances exceeded verified emissions by roughly 60Mt CO_2. This figure only reflects allowances allocated for *free* to *existing* installations. Not included are allowances that were auctioned or part of member states' new entrants' reserves. The quantity of excess allowances during Phase I was therefore even higher. When data on verified emissions for 2005 was released in April 2006, it became clear that allocated allowances exceeded emissions by at least 3 per cent. As a result, the price of Phase I EU Allowances (EUAs) crashed immediately and eventually fell below €1 per tonne of CO_2 in 2007.[12]

The extent of cap mismatch in regard to emissions varied from country to country. From 2005 to 2006, only five countries faced a net shortage of allowances: UK (−83Mt CO_2), Italy (−33Mt), Spain (−25Mt), Ireland (−6Mt), and Austria (−1Mt) (Convery et al, 2008). The net position is here defined as the difference between allocation and emissions balance. In contrast, in 11 countries, 8 of which were new member states, allocations exceeded emissions by more than 10 per cent. The differences in allowance allocation had distributional impacts, transferring money from countries with tight caps to countries with lax caps. Given the overall surplus in Eastern Europe, Convery et al (2008) estimate that facilities in Western countries acquired allowances for 41Mt CO_2, worth €700 million.

As a result of the over-allocation in Phase I, the European Commission tightened the carbon constraint for Phase II. NAPs were downsized by the Commission, in some cases to a considerable extent. In relative terms, the Baltic States had to reduce their proposed caps by the highest percentages: Estonia −48 per cent, Latvia −56 per cent and Lithuania −47 per cent. In absolute terms, the caps for Poland (−76Mt CO_2), Germany (−29Mt) and Czech Republic (−15Mt) were cut most. Overall, the Commission fixed the EU-wide cap at 2.08 billion tonnes of CO_2 annually for the period of 2008–2012. That is 9.5 per cent lower than the cap in Phase I.

Setting the post-2012 regime of the EU ETS, the EU adopted a revised EU ETS Directive in December 2008 that attempts to incorporate many of the lessons learnt during Phases I and II (EC, 2008).[13] The directive for Phase III (2013–2020) includes several changes. Total EU trading sector emissions in 2020 will be reduced by 21 per cent below 2005 levels, resulting in a cap of 1720 million tonnes CO_2.[14] To arrive at this quantity, the centrally set EU-wide cap, which replaces the 27 NAPs, decreases annually by 1.74 per cent from 2013 onwards (taking 2010 as basis year). This linear reduction factor is set to apply beyond the end of the third trading period and may be reviewed by 2025 at the latest.

The scheme's coverage will be extended to new sectors and new gases. It will cover CO_2 from emissions from aviation, petrochemicals, ammonia and aluminium, as well as N_2O emissions from the production of nitric, adipic and glyoxylic acids, and perfluorocarbons from the aluminium sector. Road transport and shipping remain excluded, although the latter is likely to be included at a later stage. Agriculture and forestry are also left out. To lower administrative costs, smaller installations, emitting under 10,000t CO_2 per year, will be allowed to opt out from the ETS, provided that alternative reduction measures are put in place. Industrial GHGs prevented from entering the atmosphere through the use of so-called CCS technology are to be treated as not emitted under the new scheme.

Allowance allocation

The current primary allocation method in the EU ETS is to grandfather allowances to sources for free. Most member states made no or only little use of the possibility of auctioning up to 5 per cent of allowances in Phase I and 10 per cent in Phase II. Only four countries used auctioning in Phase I, accounting for 0.13 per cent of total allowances. More allowances are being auctioned in Phase II, though the expected quantity is still well below the 10 per cent limit. Grandfathering was mainly based on historical emissions although benchmarking had been strongly advocated. Convery et al (2008) assume that governments decided against benchmarks because they would have resulted in allocations too far below recent emissions to gain stakeholder acceptance.[15] The use of benchmarks has spread in Phase II, mainly in the power sector, but they are typically fuel specific, that is, different for power plants generating electricity from natural gas and coal.

The power sector has been allocated almost all of the aggregate ETS sectors' emission reduction burden in Phase I, resulting in a net short position in emission allowances (Convery et al, 2008). A sector's net position is here defined as the difference between gross long (allowances received) and gross short (actual emissions) position. The industrial sector, in particular iron and steel, was long in allowances. The privileged treatment of industry can be explained by the general assumption of the sector's relatively lower abatement potential and exposure to non-EU competition.

All member states created reserves that grandfathered allowances to new entrants. Most required closed installations to forfeit post-closure allowances. This measure was adopted for political reasons. The EU as an investment location should not be disadvantaged and the incentive to close facilities and move production elsewhere should be eliminated (Convery et al, 2008). The main effect, however, was to preserve pre-policy incentives to invest in existing (i.e. relatively polluting) facilities. Even though the perverse incentives of this provision were widely recognized, it was apparently not possible to resist political demands.

The decision to grandfather permits generated substantial windfall profits (i.e. additional revenues) for incumbent firms, especially in the power sector. Auctioning allowances would recover the scarcity rent from private companies for public use without impacting electricity prices relative to the free allocation case. On these grounds, auctioning will be the principal allocation method in Phase III. The power sector will face full auctioning of permits from 2013

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onwards (this may be delayed until 2020 in some Eastern European countries), while auctioning in all other sectors is to be phased in incrementally, starting with 20 per cent in 2013 and reaching 100 per cent auctioning by 2025 at the latest. Industries that are at risk of carbon leakage may be exempted from this rule. For these industries, free allocation will be based on sector-specific benchmarks. Benchmarks will be based on the top 10 per cent most efficient facilities in the relevant sector. The European Commission determined in December 2009 which sectors are exposed to the risk of carbon leakage.

The total quantity of allowances that member states will auction will be determined at the EU level, but auctions will be organized at country level. The Directive foresees that 88 per cent of the total quantity of allowances to be auctioned will be distributed according to the relative share of 2005 emissions in the EU ETS. For reasons of 'fairness and solidarity', 12 per cent of the total quantity of allowances to be auctioned will be redistributed to poorer member states.

Did abatement occur?

In light of the excess supply of allowances in Phase I, it appears questionable whether the EU ETS was able to encourage CO_2 abatement. For the year 2007, when the EUA price dropped to a negligible level, the answer to this question is probably negative. Prior to this period, however, the EUA price consistently stayed above €6/t CO_2 and abatement should have occurred. Ellerman and Buchner (2008) estimate that for 2005 and 2006, the EU ETS reduced EU-23 emissions by between 50 and 100Mt CO_2/year. This is 2.5 per cent to 5 per cent less than the emissions expected in a scenario without EU ETS. The authors argue that the finding is valid in spite of the over-allocation that effectively existed in some countries and sectors. It is not the initial allocation that causes a source to reduce emissions but the price it must pay for its emissions, even if in opportunity-cost terms. The authors, however, emphasize that this estimate should be treated with care because there exist many potential pitfalls in constructing counterfactual scenarios. A strong argument can be made that the baseline data used for establishing their scenario without EU ETS is biased. Other studies show lower estimates for induced abatement. Delarue et al (2008) constructed a computer model in order to explore the potential of short-term CO_2 abatement in the European power sector. They think that fuel switching reduced EU-23 emissions by 35–64Mt CO_2 in 2005 and 19–25Mt CO_2 in 2006, with more weight being given to the lower bounds. The European Environment Agency (EEA, 2007) expects that reductions induced by the EU ETS by 2010 will be at least 150Mt CO_2 annually in the EU-15 and contribute 3.4 percentage points to their Kyoto-target of −8 per cent relative to 1990 levels. For the new member states, estimates differ greatly depending on the method used to calculate the baseline. The mid-case scenario predicts EU-12 emission reductions of approximately 25Mt CO_2/year by 2010. Overall, the EU ETS is expected to reduce 2010 EU-27 CO_2 emissions by 200Mt CO_2/year relative to BAU.

Besides its extent, researchers also investigate the nature of abatement. Convery et al (2008) found that abatement often occurred where it was not expected. In Germany, the power sector shifted generation from higher-emitting lignite to lower-emitting hard coal. In the UK, there was a noticeable improvement in the carbon efficiency of coal-fired power plants, probably via increased use of biomass or improved energy efficiency. Analysts, however, predicted more reductions from substituting coal with natural gas. The authors argue that this did not happen in the magnitudes expected, largely because of the relatively high natural gas prices during 2005 and early 2006. In contrast, little attention was given to either the German intra-fuel switch or the improved CO_2 efficiency observed in the UK. This underlines that market instruments may trigger abatement in areas where it was not anticipated by regulators.

Price effects

Looking at the interaction of carbon and electricity prices, there is no universal answer on how the EU ETS has affected electricity prices (Reinaud, 2007). Many other factors affect generation prices such as high natural gas prices in 2005. Gas-fired power generation is increasingly the choice of investors in volatile markets due to short lead times and flexibility advantages. With gas thus being the marginal fuel for peak production, the interaction between gas and electricity prices is particularly close. The potential use of market power by electric utilities further complicates the picture. Moreover, as no data can be gathered on the bidding strategies, and the marginal supplier to the market, determining the precise level of pass-through of CO_2 prices in electricity prices is not possible. Nevertheless, several studies attempted to provide estimates of pass-through rates: Sijm et al (2006) find rates ranging from 39 to 73 per cent for Germany and the Netherlands for the period January to July 2005 and from 60 to 80 per cent for the same countries between January to December 2005. For the reasons cited above, however, these estimates need to be treated with care.

Carbon market governance

The development of the European carbon market poses new challenges to institutions and governance. Two general areas of activity can be differentiated here: the monitoring and management of carbon markets.

Monitoring is necessary to ensure that allowance markets work efficiently. In the EU ETS, more than 10 million allowances are traded weekly on average, resulting in a market worth several billion euros. The legal nature of these allowances is, however, unclear. Some countries consider them to be financial instruments whose trading is supervised by the financial service authority, while other countries consider them to be normal commodities and only their derivatives are viewed as financial instruments (EEA, 2007). In order to avoid market manipulation and inside trading, it is important to consider how to

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apply the rules for financial markets to emission allowances. Furthermore, publication of market sensitive information by the Commission and member states should be strictly and clearly regulated, since the release of market sensitive data impacts allowance prices. The same rules that regulate the dissemination of stock market sensitive information should apply here. In the future, carbon-market governance bodies might be responsible for collecting information on market transactions, analysing their impact on the economy and industrial sectors, and disclosing this information in a predictable and transparent manner.[16] They would also act to uncover and abolish market manipulations (for example by coordinating national and regional market control authorities), organize permit auctions, and administer the overall allocation process.

The second, more controversial area of activity regards the intervention in carbon markets. The Australian ETS, for instance, will feature an 'Independent Carbon Bank' that would be in charge of regulating the carbon market based on the rules provided by the legislator (Australia, 2008).[17] Depending on the specific proposal, the carbon bank may adjust the limits on the use of offsets, extend the possibility of banking/borrowing, set the interest rates for borrowed allowances, or even sell/buy additional allowances. Though these measures may provide near-term flexibility and cost relief, the long-term reduction trajectory and emission cap may be hampered. For this reason, relief measures should only be employed incrementally after in-depth consideration of market conditions and be accompanied by a clear repayment schedule for borrowed allowances. The advantages of an intervening carbon bank can be questioned. It can be argued that the existence of a carbon bank creates uncertainty over allowance quantities if it has the power to temporarily adjust caps and borrowing modalities. Carbon prices and investments in low-emission technologies become less certain than, for example, in a fixed safety-valve approach. Further, market participants might decide to delay abatement actions because they speculate on a loosening of the carbon constraint by the intervening bank. The danger also exists that borrowed quantities are cancelled by an eventual adjustment to the cap. Nevertheless, carbon banks might satisfy politicians' and investors' demands for flexible carbon constraints and therefore represent a possible measure to provide short-term cost relief. The optimal layout and operating principles of carbon banks are the subject of ongoing research.

Complementary measures

Environmental policies are often directed at internalizing externalities. The standard theory of externalities indicates that only one instrument is needed to internalize one externality. In some instances, however, multiple externalities and market failures exist, and it is very unlikely that one instrument can be used to optimally address several market failures simultaneously. Therefore, several complementary instruments are required.

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Market failures that lead to negative abatement costs were discussed above, mostly related to energy efficiency measures. It was concluded that full sectoral coverage along with complementary regulatory measures are needed to tap into this low-cost mitigation potential. Moreover, other market failures related to the specific circumstances of technological innovation exist that prevent the large-scale uptake of low-carbon technologies such as renewable energy. For this reason, it is of utmost importance to understand that even a well-designed ETS is not sufficient on its own to encourage the fundamental energy system transformation we aim for when giving carbon a price. Although many renewable energy technologies, given stringent carbon constraints, are likely to be profitable in the long term, most of them fail to attract investments because of high initial costs. This is in spite of their significant cost-reduction potential due to learning processes and economies of scale (IEA, 2000).

The first source of failure in technology markets can be found in the public good nature of knowledge. The positive side-effects of introducing new technologies are not limited to the innovating firm because technology development typically also creates benefits for others, for example in the form of knowledge spill-overs. In the energy sector in particular, technology spill-over to competitors is large and, as a result, firms underinvest in R&D (Azar and Dowlatabadi, 1999). Without additional policies addressing this externality, investments remain lower than socially desirable even in the presence of an optimal carbon price.

Second, learning and network effects introduce multiple equilibria to the energy-economy system. In the status quo, marginal additional investments in many low-carbon technologies do not achieve profitability even though a price on carbon exists because many alternative technologies are still at the beginning of their learning curves. This state is a stable equilibrium. However, once a technology reaches a higher point on the learning curve, another stable state may be reached, one with high deployment of low-carbon technologies that is cost-optimal even if R&D and learning investments are taken into account. This bistability creates a strong path-dependency and potential technology lock-in that need to be addressed, for example via feed-in tariffs or renewable energy quotas (see Box 8.1).

Third, energy infrastructure investments compete for capital with other investment opportunities, and thus will be considered attractive only if they provide satisfying returns on investment in the near term. The benefits of research in new energy technologies, however, may not be realized for two to three decades, which is beyond the planning horizons of even the most forward-looking companies (Anderson and Bird, 1992).

Fourth, large uncertainties exist that concern the future development of energy and climate policy, availability and prices of fossil fuels, as well as the speed of innovation in low-carbon technologies. Investors, being risk averse, fear the threat of stranded investments and consequently tend to delay future engagements. This is particularly damaging for many low-carbon technologies, since they are rather capital intensive and require substantial upfront expenditures.

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Box 8.1 Renewable energy support policies in Europe

Among renewable energy support policies one can distinguish between investment support (for example capital grants, tax exemptions or reductions on the purchase of goods) and operating support (for example price subsidies, green certificates, tender schemes, tax exemptions or reductions on the production of electricity). In the EU, operating support plays a far more important role and can either work through prices or quantities. Although, in theory, both price- and quantity-based instruments should yield the same economic efficiency, practical experiences with renewable support policies in Europe draw a different picture.

A study prepared for the European Commission assessed renewable electricity support policies in various EU member states and concludes that for onshore wind, for example, in 2006 well-adapted feed-in tariffs were typically more effective at a relatively lower producer profit (Ragwitz et al, 2005). In other words, it can be observed that quota systems achieved a rather low effectiveness at comparably higher profit margins. However, it should be noted that quota systems are relatively new instruments and there is at present little knowledge of how the certificate prices will develop over time. Nevertheless, quota systems generally involve a higher price uncertainty, which leads to high risk premiums and limited economic efficiency from a social-planner perspective. They also are less capable of differentiating between technologies. For these and other reasons, feed-in tariffs, though they too have disadvantages, achieved better results in promoting renewable energies in Europe than quota-based policies.

Overall, none of these market failures is adequately addressed by an emissions trading system alone. Complementary measures and policies such as feed-in tariffs or quotas for renewable energy sources are required to support the transformation towards a low-carbon economy. At the same time, existing subsidies for unsustainable and carbon-intensive energy sources such as coal need to be abolished as they represent negative carbon prices. This may free up resources that can then be spent on low-carbon technologies. In some circumstances, complementary measures may be transitional because although they may be necessary to address a specific failure in the short to medium terms, they are not expected to be helpful in the longer term. They can support and drive research, development and demonstration of new technologies where the investors are unable to capture the full benefits of their investment and address other market failures, such as non-price barriers.

Conclusion

Emissions trading is expanding worldwide. Cap-and-trade systems effectively establish a price on carbon by setting an absolute quantitative limit for GHG emissions. The allowance cap determines the carbon price and the instrument's environmental effectiveness. But uncertainty over the carbon price makes it difficult to estimate the policy's economic cost. Trading of permits enables participants to reduce emissions where this is cheapest and leads to the formation of a price signal reflecting marginal abatement costs across covered installations. Looking at the EU ETS experience, one can draw several conclusions in regard to general ETS design:

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ETS coverage, both in terms of sectors and gases, should be as broad as possible in order to maximize market liquidity and the range of potential abatement options. Sectors with highly dispersed emission sources (for example transport) should be covered upstream in order to keep transaction costs low. Sources that are difficult to measure and monitor (for example agriculture) should be excluded from coverage to safeguard the cap's environmental integrity. In such sectors, domestic offset credits may be used to smooth the transition towards broader trading regimes by allowing experience learning with monitoring emissions. Setting and communicating long-term trajectories for cap and coverage is of paramount importance for enhancing predictability and investor confidence.

The allocation of allowances affects the policy's distributional impact and cost effectiveness. The EU ETS allocation process demonstrated that free allocation can significantly distort incentives but may ease the transition to emissions trading. Increasing the use of auctioning is likely to generate benefits in terms of cost effectiveness, distribution and public finances. Allocation procedures can be used for redistributing wealth to underprivileged regions. New entrants' provisions should be carefully balanced to set reasonable incentives while avoiding shielding incumbents against new competitors. Different allocation methods can apply to different sectors, mirroring industries' diverse vulnerabilities to carbon pricing and international competition.

Banking provisions can provide significant temporal flexibility, reducing cost and volatility. The benefit of borrowing provisions is more ambiguous as concerns over time-consistency exist. No existing trading system allows for borrowing and further research needs to clarify the conditions under which it might represent a sensible design choice.

Price ceilings limit the potential range of allowance prices. On the one hand, this reduces cost uncertainty while, on the other hand, it can diminish the incentive for investors to develop low-carbon technologies. In addition, price bounds complicate linking up with other trading systems.

MRV of emissions is critical for ensuring a trading system's transparency, integrity and credibility. Reliable historic data are essential for determining caps at appropriate levels. Stringent verification rules can avoid market-distorting ex post corrections and enhance investor trust in carbon markets. Institutions need to be put in place that oversee the carbon market and ensure that it is functioning efficiently.

Overall, carbon pricing alone is not enough to mitigate climate change. The standard theory of externalities indicates that only one instrument is needed to internalize one externality. In some instances, however, multiple externalities and market failures exist, and it is very unlikely that one instrument can be used to optimally address several market failures simultaneously. Policy-makers need to implement complementary measures that address other market failures such as underinvestment in technological innovation.

Appendix

Table 8.1 *Comparison of different cap-and-trade programmes*

	EU ETS	*RGGI*	*California*	*Australia*	*New Zealand*
Implementation Stage	Up and running since January 2005	Legislation process in progress	2007 Proposal by MAC expert commission	2007 Discussion paper by PM Task Group expert commission	2007 Final Decisions on the core design and on detailed design features on the government level
(Envisaged) Start Date	1 January 2005	1 January 2009	1 January 2012	Scheduled in 2011 or 2012	Stage 1: 1 Jan 2008 Stage 2: 1 Jan 2009 Stage 3: 1 Jan 2010 Stage 4: 1 Jan 2013
Ratification of Kyoto Protocol	Yes	No	No	Yes	Yes
Participating Sub-regions	27 EU Member States	10 US states: Connecticut, Delaware, New Jersey, New York, Maine, Maryland, Massachusetts, New Hampshire, Rhode Island, Vermont	California	Australian Commonwealth	New Zealand
Regulated Sectors	Electricity, refining, iron and steel, cement, glass, ceramics, pulp and paper	Electricity generating facilities ⩾ 25MW primarily fired by fossil fuels (coal, natural gas, oil), feeding more than 10% of their generated electricity into the grid	Pr 1:18 electricity, refining, cement, other processes, non-CO_2 Pr 2: like 1, + transport Pr 3 & 4: like 2, + small industry, commercial, residential sources	Electricity and industrial processes emitting more than 25kt CO_2-eq per annum, transport, waste to be discussed	S1: Forestry S2: Liquid fossil fuels S3: Stationary energy; industrial processes S4: Agriculture; waste
Regulated Emissions	CO_2 only	CO_2 only	CO_2, N_2O, HFCs, PFCs and SF_6	CO_2, CH_4, N_2O, HFCs, PFCs and SF6	CO_2, CH_4, N_2O, HFCs, PFCs and SF_6
Point of Regulation	Downstream	Downstream	Upstream and downstream	Upstream and downstream	Upsteam and downstream

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Table 8.1 *continued*

	EU ETS	*RGGI*	*California*	*Australia*	*New Zealand*
Covered Emissions (Mt CO_2eq)	~2.000Mt	149Mt (2003)	Pr1: 193Mt Pr2: 356Mt Pr3&4: 409Mt	~ 300Mt (according to calculations by PM Task Group)	S1: 21.8Mt (expected for the period 2008–2012) S2: 15Mt (2005) S3: 22.8Mt (2005) S4: 39.2Mt (2005) Total: 77.6Mt (excluding S1)
Share of Economy-wide Emissions (CO_2eq)		~ 24%	Pr1: 39% Pr2: 72% Pr3&4: 83%	~55%	S1: N/A S2: 19% S3: 30% S4: 51%
Number of Covered Entities	~10.000	~630	Pr1: ~450 Pr2: ~480 Pr3: ~490 Pr4: ~150	~ 900	S1: ⩾ 1000 S2: ~5 S3: ~ 80 (~45 stationary energy; ~ 35 industrial processes) S4: ⩾ 35 agriculture (Point of Regulation for agriculture not decided); ~ 60 waste
Total Emissions (incl. non-energy emissions) in Mt CO_2-eq	4979.4 (in 2004, EU25)	624.9 (in 2003)	494.3 (in 2004)	525.4 (in 2005)	77.2 (in 2005) + deforestation
Energy Mix by Sectors	Industry: 28% Households: 41% Transport: 31%	Industry: 17.5% Commercial: 25.9% Households: 27.5% Transport: 29%	Industry: 23% Commercial: 18% Households: 18% Transport: 40%	Industry + Commercial: 50% Households: 30% Transport: 20%	Industry: 30% Commercial: 9% Households: 13% Transport: 44% Agriculture: 4%
Energy Mix by Fuel	Oil: 37% Solid Fuels: 18% Natural Gas: 24% Nuclear: 15% Renewables: 6%	Petroleum products: 48.9% Coal: 8.6% Natural Gas: 23.2% Nuclear: 11.7% Renewables: 7.1%	Petroleum products: 46% Coal: 8% Natural Gas: 29.5% Nuclear: 5% Renewables: 11.5%	Oil: 35% Coal: 41% Natural Gas: 19% Nuclear: - Renewables: 5%	Oil: 38% Coal: 13% Natural Gas: 20% Nuclear: - Renewables: 28%

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Historical Emission Trends	EU25: 8% below 1990 levels in 2003; EU15: 1.7% below 1990 levels in 2003	+7.4% during 1990–2003 period	+14.3% during 1990–2004 period	+4.5% during 1990–2005 period	+23.4% during 1990–2005 period
Future Projections	4.7% above 1990 levels in 2030	12% above 1990 levels in 2019 (electricity only)	40% increase in 1990–2020 period	27% above 1990 levels in 2020	30% above 2005 levels in 2030
Reduction Goals	- 8% below 1990 levels in 2008–2012 period - 20% (or 30%) below 1990 levels in 2020 - 60%-80% below 1990 levels by 2050	- 2009 cap: 5% above 2005 levels, will remain until 2015 - 10% reduction below this cap by 2019	- 2000 levels in 2010 - 1990 levels in 2020 - 80% reduction below 1990 levels by 2050	8% above 1990 levels in 2008–2012 period	Carbon neutrality: - Electricity by 2025 - Stationary energy by 2030 - Transport by 2040
Unit of Measurement	1 metric tonne CO_2-eq	1 short tonne CO_2-eq (1 short tonne equals 0.90718474 metric tonnes)	1 metric tonne CO_2-eq	1 metric tonne CO_2-eq	1 metric tonne CO_2-eq
Cap	Future levels not specified. Bottom-up emergence of cap through NAP negotiations. Plans for centrally set cap	Annual cap of 170.6Mt CO_2 between 2009 and 2014; annual reduction of 2.5% between 2015 and 2018	Not specified	Not specified	To be linked to NZ commitments under the Kyoto protocol (309.5Mt in 2012) and an international post-2012 regime, respectively
Allocation Method	Grandfathering, benchmarking, maximum 10% auctioning	Auctioning minimum 25%; decision over remaining 75% left to individual states	Auctioning and benchmarking	Free allocation and auctioning	S1: Free allocation S2: Auctioning S3: Some free allocation for industrial processes S4: 90% free allocation for agriculture
Banking	From 2nd period on: Unlimited	Unlimited	Unlimited	Limit proposed as long as price cap applies	Unlimited
Borrowing	Rejected	Rejected	Rejected	Rejected	Rejected
Trading Period	3 years in first period, 5 years in second	3 years	No specification	10 years	2008/9/10–2012 2013–2020
Compliance Period	1 year	3 years	3 years recommended	1 year recommended	1 year (S1: initially 2 years)

Table 8.1 *continued*

	EU ETS	*RGGI*	*California*	*Australia*	*New Zealand*
Price Cap	Rejected	Two-stage safety valve arrangement: 'Credits Trigger Event' if spot price for emissions exceeds \$7 for a period over 12 months; 'Safety Valve Trigger Event' if spot price for emissions exceeds \$10 for a period over 12 months	Rejected	Proposed; further specifications to be discussed	Rejected in principal, but considered if no international climate policy agreement post-2012
Price Floor	No	Yes	Encouraged for consideration	No	No
Penalty System	Delivery of the non-delivered allowances + €100 penalty per tonne (2008–2012)	Three times the non-delivered certificates to be delivered at next compliance date	Not specified, but non-delivery shall be made up + penalty	Emissions fee proposed setting a price cap	Delivery of the non-delivered allowances (can be extended to two times of the non-delivered allowances) + NZ\$30–60 penalty per tonne
MRV	Updated in 2007	Guidelines for continuing measurement based on CRF 40 Part 75 (Acid Rain Program regulation) that demands a maximum uncertainty of 10%. Verification by regulating authority	To be developed, building on existing MRV infrastructure of California Climate Action Registry	To be developed	To be developed

Registry	Community Independent Transaction Log overseeing communications between national registries	No common registry for RGGI	To be developed, building on existing CCAR infrastructure	Under development; rules clarified under the Australia National Greenhouse and Energy Reporting Act in legislation since September 2007	To be developed building on existing infrastructure established under the Climate Change Response Act 2002
Domestic Credit Programme	No	Credits are accepted from programmes in RGGI states or any other US state or jurisdiction	To be developed using RGGI experience; programmatic approach proposed	To be developed especially focusing on forestry and agriculture	Discussed
Eligibility of CDM/JI	Yes	Generally no; in case of a safety trigger event credit allowances may be awarded for the retirement of allowances or credits from international trading programmes	Most MAC members recommend eligibility of CDM	Yes	Yes + assigned amount units (AAUs)
Import Quota for Credits	Varying from country to country according to set of criteria. Average (weighted) EU quota for CDM/JI import in trading period II: 13.4%	3.3% of a facility's emissions can be covered by credits; the number rises up to 10% in case of a safety valve trigger event	Most MAC members recommend unrestricted credit import	No	No

Source: Adapted from Flachsland et al (2008)

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Table 8.2 *Revision of EU ETS Directive in perspective*

	2005–2007 (*Directive 2003/87/CE and 2004/101/CE*)	*2008–2012* (*Directive 2003/87/CE and 2004/101/CE*)	*Proposal for 2013–2020* *Draft COM(2008)16 final*
Perimeter	EU25 + Bulgaria and Romania in 2007	EU27 + Norway, Iceland and Liechtenstein	EU27 + Norway, Iceland and Liechtenstein Possibility of linking the EU ETS to other national or regional cap-and-trade systems
	CO_2 only	CO_2 ; + N_2O in France and the Netherlands	CO_2 + N_2O and PFC
	50% of allowances allocated to electric power producers, 50% of allowances allocated to other combustion activities (including refining), cement, steel, glass, ceramics and paper producers	Same sectors as 2005–2007 + aviation beginning in 2012	Same sectors as 2008–2012 + aluminium and ammonia production. Discussion of the inclusion of maritime shipping
Capping of emissions	National (NAP validated by the European Commission)		European cap, with auctioning quotas distributed among states on the basis of their historical emissions in 2005 and their economic situation
	2300 million allowances annually (including reserves of 70Mt/year)	2100 million allowances annually (including reserves of 120 Mt/year, non-constant perimeter)	Without an international agreement: −20% over the period 2020: 1720Mt 2013: 1870Mt With an international agreement: 2013: 1970Mt −30% over the period 2020: 1375Mt
General allocation procedures	Free for at least 95% of allowances. Less than 1% of the allowances were effectively auctioned (principally Denmark, Hungary and Ireland).	Free for at least 90% of allowances. Auctions planned for at least 3% of the allowances (principally Germany, UK, Netherlands and Hungary)	Auctioning of all allowances beginning in 2013 in the electric power generation sector. Linearly increasing auctioning in the other sectors (beginning at 20% in 2013 and reaching 100% in 2020)
	⇨ Essentially free allowances		⇨ Essentially auctioned allowances
Allocation to new entrants	Reserves of allowances and national allocation method 70Mt/year	120Mt/year	European reserve and allocation method on the basis of harmonized rules
Kyoto carbon credits (CER and ERU)	Permitted, but not used due to over-allocation	Utilization limited to 13.5% of the allocation on average (from 0 to 20%, depending on the country) Total maximum import of 1.4 billion credits, i.e. 280Mt/year	Without an international agreement: Possible use of unrestored surplus of 2008–2012 credits for compliance With an international agreement: 2008–2012 surplus + volume equal to 50% of the additional effort required
Other carbon credits			Possibility of creating harmonized European domestic offset projects or other types of international projects
Distortion of competition with non-EU countries	Handling by the states via the rate of effort applied to each sector in the NAP. Principal effort by the electric power generation sector, which is subject to relatively little international competition.		If necessary, the Commission will propose the continuation of free allowances in certain sectors, or some other mechanism of the border tax adjustment type
Registry	National registries + European registry (CITL)	European registry only	

Source: Adapted from Mission Climat (2008)

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Notes

1 The relative merits and drawbacks of both types are subject of an ongoing academic debate (see for example Quirion, 2005; Sue Wing et al, 2006; Newell and Pizer, 2006).
2 The EU ETS covers more than 10,000 different installations, 7 per cent of which account for 60 per cent of total emissions. In contrast, the 1400 smallest sources account for less than 0.14 per cent of emissions (EC, 2008). Reducing the number of covered sources can cut a programme's cost without significantly affecting its environmental effectiveness, but only if there is no leakage from larger to smaller sources. Alternatively, upstream coverage automatically includes all sources within a sector.
3 Because the amount of CO_2 emitted from burning a unit of coal, oil or gas is largely invariant to the process in which it is oxidized, emission factors can be calculated easily on the level of fossil primary energy carriers. Indeed, observing coal use and calculating emissions from it rather than observing emissions in the flue stack is the standard procedure for calculating emissions, for example for large-scale users of coal in the EU ETS, and also in national emissions inventories under the UNFCCC.
4 See for example Wara and Victor (2008) or Schneider (2007), who provide a critical analysis of the CDM's environmental performance.
5 The RGGI system uses short tonnes.
6 Garnaut (2008) argues that good governance and stringent law enforcement can avoid this problem. However, the assumption that governance is good and law enforcement is stringent at all times and everywhere can be questioned. After all, if the political agenda of a legislating body is changed following general elections, former decisions on caps etc. may be challenged and eventually altered.
7 For example, the US 'Bill to Reduce the Economic Impact of Climate Legislation' proposes the establishment of a 'Carbon Market Efficiency Board', which would intervene in a future US carbon market when allowance prices exceed a certain threshold. The first measure is to expand companies' ability to borrow permits. The second measure, to be used if high prices are not relieved by the first measure, is to add permits to the market. This temporary increase would be compensated for by reducing available allowances in a later year.
8 A special case of borrowing represents intra-period borrowing. This is the case in the EU ETS, where facilities receive allocations for two compliance years prior to the first actual compliance date. That is, until the very last compliance year in a trading period, there are always two allocation tranches in the market at the date of compliance. Intra-period borrowing does not dilute environmental effectiveness.
9 Several authors have concluded that tradable permit programmes may be less appropriate for developing countries due to their lack of appropriate market or enforcement institutions (Blackman and Harrington, 2000; Bell and Russell, 2002). For China, Wang et al (2004) suggest that a strengthened monitoring and enforcement capacity would be required to implement emissions trading.
10 Reinaud (2008) correctly points out that most discussions of competitiveness effects neglect industries that benefit from climate policy. This is fallacious as climate policy potentially enhances research, development and deployment of lower-emitting technologies and also offers positive competitiveness and employment effects.

11 The following discussion is largely based on the findings provided by Convery et al (2008).
12 Note that in the presence of banking between trading periods, the price would not have crashed to zero since over-allocated allowances could have been used in Phase II or Phase III. However, banking was not allowed. This illustrates the role of banking in containing price volatility.
13 Table 8.2 in the appendix to this chapter summarizes the main innovations of the draft directive and puts them into perspective with the current trading regime. Their final validation will depend on the results of the negotiations among the European Commission, Council and Parliament. Political consensus was achieved in December 2008.
14 The numbers are based on the scope of the ETS as applicable in Phase II.
15 In the EU ETS, choice of allocation method rests with national governments.
16 Several companies have emerged providing market data, projections and background information on the EU ETS (see for example. www.pointcarbon.com).
17 The Carbon Market Efficiency Board mentioned in Footnote 7 is another example of a carbon market oversight institution.
18 The Market Advisory Committee has proposed four programme options (abbreviated Pr here) with differing coverage and points of regulation.

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