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Carbon Markets: Past, Present, and Future

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Abstract

Carbon markets are substantial and they are expanding. There are many lessons from experiences over the past eight years: fewer free allowances, better management of market-sensitive information, and a recognition that trading systems require adjustments that have consequences for market participants and market confidence. Moreover, the emerging international architecture features separate emissions trading systems serving distinct jurisdictions. These programs are complemented by a variety of other types of policies alongside the carbon markets. This sits in sharp contrast to the integrated global trading architecture envisioned 15 years ago by the designers of the Kyoto Protocol and raises a suite of new questions. In this new architecture, jurisdictions with emissions trading have to decide how, whether, and when to link with one another, and policymakers overseeing carbon markets must confront how to measure the comparability of efforts among markets and relative to a variety of other policy approaches.

Key Words: carbon market, tradable permit, allowance, climate change, greenhouse gas

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1. Why Carbon Markets?

The 1980s and 1990s witnessed a growing awareness of climate change risks and the associated need to reduce greenhouse gas emissions. Beginning with the first World Climate Conference in 1979, attention culminated with the signing of the UN Framework Convention on Climate Change (UNFCCC) at the 1992 Earth Summit in Rio de Janeiro. In that agreement, 166 (now 194) nations acknowledged the need to limit the accumulation of greenhouse gases (GHGs) in the atmosphere to a level that will "prevent dangerous anthropogenic interference with the climate system."[1]

During that same period, emissions trading began emerging as a practical and increasingly popular policy tool to address pollution control, particularly with the successful phasedown of lead in gasoline and creation of the acid rain trading program in the United States (Tietenberg 1985; Stavins 1998). On the international scene, the Montreal Protocol on Substances that Deplete the Ozone Layer, signed in 1987, laid clear groundwork for the idea of targets and timetables for emissions levels in different countries, and it included a limited amount of emissions trading. It is therefore not surprising that there was considerable enthusiasm for using this tool to address climate change as countries grappled with how to design the 1992 UNFCCC and then meet its objectives in the 1990s. Indeed, advocacy for international GHG emissions trading began in the late 1980s and early 1990s: the United States initially promoted it in the UNFCCC treaty negotiations, and the idea of "joint implementation" as an informal version of emissions trading ultimately appeared in the UNFCCC (Wiener 2001).[2]

This enthusiasm for emissions trading ultimately overcame various objections and led to the signing of the Kyoto Protocol in 1997.[3] The Kyoto Protocol was the first vehicle for emissions trading in greenhouse gases—or what we will call carbon markets.[4] Kyoto set up a system of emissions limits for a basket of six GHGs for developed countries, mechanisms for those developed countries to trade their emissions limits, and mechanisms for developed countries to offset their emissions by financing emissions reductions in developing countries. While the Kyoto Protocol itself has led to a very small number of trades directly among countries, the European Union and a variety of other jurisdictions have since pursued emissions trading to reduce their regional GHG emissions. Carbon markets are now the largest class of environmental or emissions trading markets in the world in terms of both volume and market value, by a very wide margin.

But how effective has emissions trading been at addressing climate change? What are the distinguishing features of trading in emissions of greenhouse gases compared to, for example, conventional air or water pollution? What have we learned as carbon markets have been

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[1] See Article 2 of the UN Framework Convention on Climate Change.

[2] Joint implementation refers to one nation financing (partly or in whole) an emissions reduction project in another nation.

[3] Objections included an early focus on emissions taxes in the European Union (Barrett 1998) as well as more general issues related to emissions trading (e.g., Sandel 1997).

[4] We use the term "carbon market" because carbon dioxide is the dominant gas in terms of its overall contribution to global warming and because the units of trade are denominated in terms of carbon dioxide equivalent.

designed, implemented, and operated? The purpose of this paper is to answer these questions and highlight new emerging issues that now need to be confronted. After eight years of carbon market experience following the creation of the EU Emissions Trading System in 2005 (see Figure 1)—experience that includes growing market volumes, market value, and emissions coverage—at least part of that answer must be, yes, carbon markets can work effectively. However, we have learned a lot, and there are important lessons for current and future policymakers, analysts, and researchers.

The next section provides an overview of the normative theory of carbon market design, beginning with a description of some of the relevant attributes of GHGs. Section three discusses actual experience with existing and proposed emissions trading program design, while section four describes and draws lessons from operational experience in those markets. Section five looks forward and describes key issues facing emissions trading policies.

2. Normative Theory of Policy Design for Carbon Markets

There are a number of distinguishing attributes of the climate problem that are relevant to the design of carbon markets and that differentiate them from most other emission or resource markets. The first is the global nature of the climate problem. GHGs are one of the few examples of a global, uniformly mixed pollutant: GHG emissions throughout the world have the same consequences regardless of where they are emitted.[5] Put another way, GHG emissions have exactly the same externality properties across countries that many conventional emissions have within a local jurisdiction. It is insufficient to control the GHG emissions from a particular source or region to reduce the risks to that region. This feature of the climate problem requires that, ultimately, an internationally coordinated approach must be taken, with implications for carbon markets that make them unusual if not unique among environmental markets.

Second, GHGs are long-lived stock pollutants, remaining in the atmosphere on the order of decades to centuries. It is the accumulated global atmospheric concentration of GHGs, rather than an amount emitted at a particular point in time, that is linked to global warming and climatic change. Along the same lines, the capital stock that produces, distributes, and consumes GHGs also tends to be long-lived; cars and major appliances can last over a decade, while power plants and buildings last several decades. It is therefore important to keep in mind a long-term global perspective when addressing the climate problem, even when considering particular near-term regional policies.

Third, while carbon dioxide is the major GHG, there are numerous GHGs of varying potency (quantified in terms of the amount of heat energy trapped by a given amount of each gas) and longevity, from methane, with a lifetime of 12 years, to sulfur hexafluoride, with a lifetime of 3,200 years.

Fourth, GHGs are pervasive in the economy, rather than being identified with a particular set of sources, sectors, or technologies. Carbon dioxide, the dominant GHG, is a fundamental product of the combustion of fossil fuels (coal, oil, and natural gas) for energy production.

[5] Ozone-depleting substances are one other example.

Energy is used everywhere, and fossil fuels are the source of over 80 percent of U.S. and global energy consumption. As a consequence, the potential market size is much larger than other existing environmental markets. For example, in 2008, fossil fuel combustion accounted for about 30 billion metric metric tons of global carbon dioxide emissions (Boden *et al.* 2011). The market value of one year of allowances for these emissions at $10 per metric ton would be $300 billion; at $25 per metric ton it would be $750 billion. For higher allowance prices, or when aggregated across several vintages of allowances, the value is easily in the trillions of dollars. As another example, the estimated market value of allowances that would have been created, just in the United States, by proposed Senate legislation in the 110th Congress was $6–7 trillion (Samuelsohn 2008).

A related point is that while stabilization of GHG concentrations in the atmosphere eventually implies driving net greenhouse gas emissions to near zero, the foreseeable future likely entails significant continued emissions. The pervasive economic role of fossil energy use coupled with limited alternatives and continued global economic growth means that GHG emissions will not be reduced in the same way that ozone-depleting substances were virtually eliminated in the 1980s and 1990s (Sunstein 2007). In scenarios with limited to moderate emissions reductions, total mitigation costs tend to be a small fraction of the allowance value (Burtraw & Evans 2009). This leads to different distributional consequences than more typical situations where mitigation is a significant share of total emissions.

A fifth important feature is that options exist for offsetting GHG emissions, thereby negating or at least significantly reducing their impact on the atmosphere. Generally speaking, offsets include any approved methods for reducing emissions or removing GHGs directly from the atmosphere (e.g., through forestry) that are not otherwise covered under a particular cap. In principle, offsets can occur within or outside of the jurisdiction covered by an emissions cap. On a related note, carbon capture and storage (CCS) technologies offer an end-of-pipe solution, avoiding atmospheric emissions by capturing carbon dioxide from process or combustion gases and storing it underground. While end-of-pipe approaches are quite common in conventional air and water pollution control, CCS is considered a relatively advanced approach in the climate context and is still in the development stage. Nonetheless, as is the case with offsets, well-designed policy would need a mechanism for crediting CCS.

Given the degree of complexity and long timeframes inherent in the climate problem, uncertainty along several dimensions—including the effectiveness of different approaches, the cost of mitigation, the evolution of technology, and the climate risks themselves—tends to be an overarching and pervasive aspect of policy analysis, construction, and implementation.

When placed in the context of the economic paradigm, the above attributes of the climate problem have a number of implications for carbon market design. We discuss the most important of these in this section, including the importance of compliance flexibility for cost-effectiveness; the significance of benefit and cost features for efficient instrument design; and the link between allowance allocation, government revenue and use, distributional impacts, and international competition. For further detail on several of these design issues, Aldy et al. (2010) provide a thorough review of the literature on designing climate mitigation policy.

2.1 Cost-effectiveness, Comprehensiveness, and Flexibility

Cost-effectiveness, or achieving a given aggregate GHG mitigation target at the lowest possible cost, tends to be a top-tier concern for market-based policy design. To theoretically enable cost-effectiveness (although not empirically guarantee it), the above characteristics tend to point to the design of carbon markets that have comprehensive coverage (i.e., across all gases, sectors, sources, and technologies), can take advantage of mitigation opportunities at a global scale, and have a long-term time horizon and structure that supports integrated decisionmaking across time. This is sometimes referred to as "what, where, and when flexibility," which underpins a normative tendency toward comprehensiveness and minimal barriers to trading opportunities across space, time, and mitigation activity.

"What flexibility" relates to the comprehensiveness of any carbon market, with theory guiding design toward inclusion of as many types of GHGs (e.g., carbon dioxide, methane, nitrous oxide, and fluorinated compounds), sectors (e.g., electricity, transport, industry, and agriculture), and technologies (e.g., fuel switching, CCS, and forestry and other biosequestration) as is feasible. One caveat is that it is important to consider what happens outside of a global trading system when only some jurisdictions are pursuing emissions trading. In particular, "leakage" can occur when regulation within one jurisdiction leads to increased imports from another, unregulated jurisdiction—a phenomenon that may apply to a variety of environmental policies, not simply carbon markets. Leakage tends to be concentrated in energy-intensive, trade-sensitive sectors and has both emissions and economic consequences as economic activity shifts across regions. There are a variety of ways to address these concerns (Fischer & Fox 2009).

"Where flexibility" speaks to the significant variation in the costs of GHG mitigation across regions and countries. When focused on designing a carbon market for any specific region, it is therefore important to consider the extent to which allowances and/or offsets from other regions will be permitted to be used for compliance. So long as the validity of such allowances can be established, cost-effectiveness would tend to point toward open trade in allowances across jurisdictions. However, this type of linkage can face practical and political hurdles, in part due to the associated resource transfers, as we will discuss below (see Section 5.1).

Finally, given the stock pollutant nature of GHGs, emissions on any particular day, month, or year are not consequential for climate impact. As a result, allowing "when flexibility" through banking or borrowing of allowances across time tends to increase cost-effectiveness without additional harm to the climate. So long as the full range of mitigation options across space and time is available to the market system, trading will in principal lead to a cost-effective allocation of pollution control actions through the elimination of opportunities for arbitrage.

2.2 Efficiency and Instrument Design

In addition to cost-effectiveness, which implies equalization of marginal pollution control costs, overall economic efficiency requires balancing the marginal benefits and marginal costs of GHG mitigation. One of the advantages of a market-based cap-and-trade approach to pollution control is that—for a given emissions target—it does not require any ex-ante knowledge of pollution control costs or benefits. However, if there is interest in balancing the marginal cost and benefits of the policy, economic information is required for both sides of this ledger.

This raises additional issues related to estimating the costs of mitigation, the monetized value of climate damages (Interagency 2009; National Research Council 2010), and the design of policy instruments that are efficient in the face of uncertainty over costs and benefits. The early economic literature on climate policy instrument choice under uncertainty (Pizer 2002; Newell & Pizer 2003) pointed to the advantage of price-based (i.e., a carbon tax) over quantity-based instruments (i.e., cap-and-trade) based on a modified Weitzman-type argument (Weitzman 1974) and assuming compliance had to be achieved on an annual basis. This earlier work also showed that price-like modifications within a cap-and-trade program—ceilings and floors on the allowance price or otherwise adjusting the cap to accommodate cost shocks—could achieve the same outcome as a carbon tax (Newell *et al.* 2005; Murray *et al.* 2009). Moreover, recent work has shown that ordinary banking provisions—which are included in virtually all emissions trading programs—may come close to the efficiency of a carbon tax without further modifications (Fell *et al.* 2012).

One of the thorny issues raised by the potential movement of allowances across time is determining the trading ratio to be applied to banked or borrowed allowances and deciding how to apply it. Theory suggests that the optimal trading ratio between periods is equal to one plus the discount rate minus the desired rate of change in permit prices (Leiby & Rubin 2001). In addition to this formula, a discount rate is required, which raises a distinct set of analytical challenges for both the estimation of damages and the rate at which the carbon price should rise (Interagency 2009; Aldy *et al.* 2010; National Research Council 2010).

Due to the long-term nature of the climate problem, incentives for technological innovation in GHG mitigation technologies are also important for efficiency (Newell 2010). With respect to carbon market design, a key issue for technology innovation and deployment is setting carbon market stringency at a level that generates the optimal incentive through the price mechanism. In the presence of innovation spillovers, one might be tempted to boost stringency beyond the level dictated simply by the climate externality. However, a more economically efficient approach is to recognize the need for more than one policy instrument to address these multiple market imperfections and include direct innovation policy (Jaffe *et al.* 2003). Instrument design in the context of multiple market imperfections and multiple policy instruments can become complex, as the optimal level of stringency of any particular instrument depends on the levels of the others (Fischer & Newell 2008).

2.3 Allowance Allocation: Distributional Impacts and Efficiency

As with other market-based programs, GHG allowances can be auctioned, allocated for free, or designed with some combination of the two. Allowance allocation has distributional and efficiency consequences, and these can be large—on the order of hundreds of billions of dollars—given the sizable economic rents at stake. As noted above, because of the relatively small ratio of mitigation to emissions in most carbon trading regulation to date, payment of these rents tends to be the dominant distributional cost.

In many key carbon-emitting sectors, we would expect competitive pressure to lead product prices to reflect carbon content, regardless of any free allocation.[6] Consequently, end users of energy would ultimately end up paying these rents. In turn, there can be significant distributional impacts to alternative free allocation approaches and formulas that distribute these rents, with implications for feasibility and equity across firms, income groups, regions, and generations. Unless utility or other regulation intervenes, free allowance allocations tend to accrue to firms and result in higher equity values, to the benefit of higher income groups and at the expense of lower income groups (Dinan & Rogers 2002).

Even without the potential equity impact of free allocation, it is fairly well established that carbon pricing (through either an auctioned allowance market or tax) disproportionately harms low-income households in developed countries (Hassett *et al.* 2009). While higher-income households tend to spend more on energy than do low-income households, they spend less as a fraction of income. A number of mechanisms have been proposed to ameliorate the regressiveness of carbon pricing, including lump-sum rebates (e.g., so-called cap-and-dividend), allocations to electricity distributors, and parallel offsetting changes to income or social security taxes (Burtraw *et al.* 2010). From an efficiency point of view, better compensation mechanisms tend to avoid lowering energy prices (to preserve appropriate incentives) and take advantage of opportunities to lower other distortionary taxes.

Absent a global trading program, free allowance allocations for energy-intensive industries are often a key mechanism to address political feasibility as well as to avoid emissions leakage for marginal producers. For moderate policies, some research on this issue has found that, in the United States, only about 15–20 percent of free allowances are needed to compensate energy intensive industries for their loss of producer surplus (Bovenberg & Goulder 2001). However, other research has shown that there is significant variation in compliance costs *within* energy intensive sectors—creating both winners and losers under a carbon market—so that it can require a several times higher fraction of allowances to compensate the losers than would be indicated by the average loss in producer surplus (Burtraw & Palmer 2008).

Efficiency issues arise principally with respect to how much revenue is raised through auctions and how much of this revenue is recycled to offset other distortionary taxes or used for other purposes (e.g., technology programs, adaptation, or energy efficiency programs). The literature tends to support the view that revenue-neutral full auctioning and the use of proceeds to lower other distortionary taxes both offer a significant potential gain in efficiency relative to free allocation or lump sum redistribution of revenues (Aldy *et al.* 2010).

There are any number of other ways that carbon allowance auction revenues could be used, and the potential need for resources to support complementary GHG mitigation efforts—such as funding energy efficiency programs and technology research, development, and demonstration—has linked these programs to carbon market design. It is not clear there is any efficiency advantage to this linkage, but from a practical point of view, the use of carbon market revenue to support other GHG mitigation programs has had appeal to policymakers and has

[6] Exceptions to such pass-through could include utilities under cost-of-service regulation (where such pass-through is prohibited) and industries facing strong international competition (and thus world prices).

shown up as a prominent feature of both proposed and implemented carbon market design. However, interactions between carbon markets and other policies aimed at reducing emissions may—under certain circumstances—result in decreased emission reductions, reduced cost-effectiveness, and/or a decrease in the market's allowance prices (Goulder & Stavins 2012).

3. Program Overview and Key Design Choices

GHG emissions trading programs have been established at the international, regional, national, and sub-national levels, as highlighted in Figure 1. Though the normative theory discussed above focuses on cost-effectiveness, regulators developing each program have had to grapple with significant stakeholder interest in many design choices. The most contentious issues have revolved around which sectors to include; how to minimize competitive losses to firms; how to limit consumer price increases; whether and how to employ emissions offsets; and the extent to and manner in which firms trade allowances.

This section explores those key questions, summarizing the initial design choices in a variety of emissions trading programs and highlighting notable design elements. Table 1 gives a broad overview of central design elements in a variety of past, present, and (for the United States) proposed carbon trading systems.

Additionally, this section touches on the institutional and political histories of major programs. Although we cannot do full justice to the broad array of political and economic choices leading to each decision, we note some key factors that affected initial design choices. While all carbon trading programs are relatively young, some have changed substantially since launching. We describe those major changes, and the market experiences that prompted them, in section four as we discuss market experience more generally.

3.1 Program Overview

3.1.1 *Kyoto Protocol*

The Kyoto Protocol, adopted in 1997, established the first non-voluntary carbon market, committing certain nations to meet GHG emissions reduction targets and establishing a framework for allowance trading across international borders. The protocol entered into force in February 2005. Thirty-seven industrialized signatories, known as Annex-I nations, are responsible for reducing emissions by specified targets (United Nations Framework Convention on Climate Change 2012b). If a nation cannot meet its target, it may either purchase allowances (called Assigned Amount Units, or AAUs) from a fellow Annex-I nation or purchase emissions offsets from projects that decrease emissions in other parts of the world.[7]

The Kyoto Protocol established two major mechanisms to offset emissions: the Clean Development Mechanism (CDM) and Joint Implementation (JI). CDM projects, which reduce emissions in developing nations, earn one Certified Emissions Reduction credit (CER) per metric ton of GHG emissions reduction, which may be purchased by nations or firms to meet their

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[7] As predicted by Hahn and Stavins (1999), the widespread trading of AAUs has generally not been effective.

obligations under the protocol. JI projects also earn one credit per metric ton of emissions reduction (called Earned Reduction Units, or ERUs) and come primarily from projects in the former Soviet Union. Both project types seek to encourage clean energy investment and learning while allowing Annex-I nations flexibility in meeting their emissions targets (United Nations Framework Convention on Climate Change 2006).

While these offset programs continue to function, the future of the Kyoto Protocol as a framework for reducing emissions is uncertain. Negotiations in Durban, South Africa, in late 2011 ended with nations agreeing to agree by 2015 on a comprehensive plan to reduce emissions that would take effect by 2020 under the broader UNFCCC. In 2012, negotiations in Doha, Qatar resulted in the extension of the Kyoto Protocol from 2013-2020. However, the United States never ratified the protocol; China (the world's largest emitter) is not required to reduce emissions under it; and Canada, Japan, and Russia have not agreed to take on additional commitments under Kyoto after 2012 (United Nations Framework Convention on Climate Change 2012b). Australia has committed to modest emissions reductions by 2020, promising to enact deeper cuts if the world's other major emitters commit to an "ambitious global deal" (United Nations Framework Convention on Climate Change 2011). New Zealand has stated that it will not make any commitments under the second round of the Kyoto Protocol, though it remains committed to its overall emission reduction goals.

These varying commitments paint an uncertain picture moving forward. Nonetheless, a variety of governments have developed market-based programs to reduce their GHG emissions within or outside of the Kyoto framework. Why these governments have chosen to act absent a strong international framework is a question we do not focus on; instead readers should see work by Aldy et al. (2007) on international climate architectures, and Victor (2008) and Nordhaus (Nordhaus & Boyer 1998) on problems with the Kyoto approach. The following section provides a review of major programs.[8]

3.1.2 *EU Emissions Trading System*

Although most European nations favored a carbon tax going into the Kyoto negotiations, the European Union has created by far the world's largest market-based system to reduce GHG emissions: the EU Emissions Trading System (EU ETS). In fact, the EU ETS began operating in 2005, three years before Kyoto's first commitment period began. The program has operated in phases, with a pilot phase from 2005 to 2007 covering the power sector and certain heavy industry, a second phase from 2008 to 2012 expanding coverage slightly, and a third phase set for 2013–2020 that adds a significant range of industrial activity along with several new gases (Ellerman *et al.* 2010; European Commission 2012a).

Under the first two phases, each of 27 EU nations (later expanded to 30) submitted National Allocation Plans (NAPs) to the European Commission.[9] Each NAP set an emissions

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[8] For more details on other proposed and existing programs, see Hood (2010).

[9] Austria, Belgium, Bulgaria, Cyprus, the Czech Republic, Denmark, Esmetric tonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, the Netherlands, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, and the United Kingdom. The program later added Iceland, Liechtenstein, and Norway.

target for covered sectors and was reviewed by the Commission, which had the power to adjust the number of allowances allocated to each EU member. The program's initial design was highly decentralized, and once the NAPs were finalized, nations had significant discretion on how to distribute emissions credits to different sectors of their economies .

The pilot phase was something of a test during which governments and the private sector could develop a better understanding of how the trading program would function as the cap tightened in future phases. NAPs were developed under tight deadlines, and the Commission's adjustments to each national proposal were hampered by shortages of time and data. Despite the difficulty of calculating business-as-usual (BAU) projections with incomplete data, the Commission developed targets based on a modest emissions reduction goal (Ellerman *et al.* 2010).

Even with modest reduction targets, member nations worried that pricing carbon could harm industrial sectors subject to international competition and raise consumer costs for domestic electricity. As a result, the vast majority of European Union Allowances (EUAs) were allocated free of charge in the pilot and the second phases. Each nation, through its NAP, determined the level and distribution of free allocation to different sectors through 2012 . NAPs also specified the number of offset credits emitters in each nation could purchase from CDM or JI projects, with limits ranging from 0 to 20 percent (European Commission 2009).

Market experience and research and analysis in the EU ETS have led to significant changes as the program enters its third phase in 2013. We discuss that experience and changes in allowance allocation, offset rules, and the ability to bank credits in section four.

3.1.3 *United States*

The United States, which signed but never ratified the Kyoto Protocol, has seen dozens of legislative efforts to address GHG emissions. Although the effectiveness of an emissions trading approach was first demonstrated on a large scale by the United States' sulfur dioxide and nitrous oxide programs under the 1990 Clean Air Act Amendments, the U.S. Congress has not passed a GHG trading program (Burtraw *et al.* 1998; Meckling 2011).

Building on a variety of past efforts beginning as early as 2003's Climate Stewardship Act (cosponsored by Senators Joseph Lieberman and John McCain), the U.S. House of Representatives narrowly passed the American Clean Energy and Security Act in 2009 (better known as the Waxman-Markey bill, after the bill's cosponsors).[10] The Waxman-Markey bill would have established an economy-wide emissions trading program, which focused initially on the power and industrial sectors, expanding after several years to include transportation, and allowing emitters to offset up to 30 percent of their caps from either U.S.-based or international offset projects (Larsen *et al.* 2009). However, political opposition doomed the legislation in the Senate, where it was never taken up (Meckling 2011). Despite the absence of a nationwide system, several regions had already put in place or developed plans to implement their own GHG trading programs.

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[10] For details on a variety of U.S. emissions trading proposals, see Aldy and Pizer (2008). For details on the 2003 McCain-Lieberman Act, see Pizer and Kopp (2003).

In 2005, seven northeastern states became the first collection of jurisdictions in the United States to agree to an emissions trading program: Connecticut, Delaware, Maine, New Hampshire, New Jersey, New York, and Vermont. Maryland joined in 2006, Massachusetts and Rhode Island joined in 2007, and New Jersey withdrew at the end of 2011. Known as the Regional Greenhouse Gas Initiative (RGGI, pronounced as “Reggie”), this program only covers large electricity generators, and nearly 100 percent of allowances are auctioned. Revenues from allowance auctions go to state governments, which are required to invest at least 25 percent of those funds toward energy efficiency or renewable energy programs. Offsets for emitters are limited to just 3.3 percent and come from projects mostly within RGGI states, though no offset projects have yet been used for compliance (Regional Greenhouse Gas Initiative Inc. 2012).

A second U.S.-based emissions trading system began in California in 2013. Arizona, California, New Mexico, Oregon, and Washington initiated the Western Climate Initiative (WCI) in 2007, later adding Utah and Montana as well as British Columbia, Ontario, Manitoba, and Quebec (we discuss these Canadian provinces below). However, no WCI states in the U.S. other than California have implemented an emissions trading program (Craig 2011). The WCI has developed detailed policy proposals to coordinate future programs implemented by its members, with an eye toward linking state programs under one umbrella.[11] California’s emissions trading program, mandated by legislation, uses provisions developed by the state’s Air Resources Board in consultation with the WCI and will auction roughly two-thirds of its allowances in 2013 . California’s program will initially cover the power sector and large industrial sources, accounting for roughly 37 percent of the state’s GHG emissions, and expand to cover 85 percent by 2015, including transportation fuels (California Code of Regulations 2011).

A final regional agreement, the Midwestern Greenhouse Gas Reduction Accord (MGGRA), was signed in 2007 by six governors. The accord, which included Illinois, Iowa, Kansas, Michigan, Minnesota, Wisconsin, and Manitoba along with several observer states, was to function like the WCI, helping its members establish programs by 2012. In 2010, MGGRA released a detailed model rule for member states to adapt and use. The proposal covered the power sector, major industrial emitters, and transportation fuel distributors, with most allowances being allocated free of charge at the discretion of member states (Midwest Greenhouse Gas Reduction Accord 2010). Despite the MGGRA’s progress, new political leadership and shifting economic priorities ended the effort in early 2011 (Volvovici 2011).

3.1.4 *Canada*

Canada ratified the Kyoto Protocol in 2002 but never passed comprehensive legislation to regulate emissions. Driven by strong economic growth and a booming natural resources sector, CO_2 emissions exceeded their targets, and Canada withdrew from the Kyoto Protocol in 2011 (Canadian Government 2011b). Many Canadian policymakers are reluctant to act on climate change without the United States taking comparable action. Nonetheless, several provinces have made independent efforts to reduce GHG emissions (Canadian Government 2011a; De Souza 2011).

[11] For details, see www.westernclimateinitiative.org and www.wci-inc.org.

Quebec is currently on track to implement a GHG emissions trading program starting in 2013 . Since nearly all of its electricity comes from non-carbon-emitting hydropower, the program will not affect most electricity generators but will apply to industrial sources (Centre for Energy 2012). Quebec developed the system using provisions developed under the WCI, allowing it to link with California (for a more detailed discussion on linking, see section 5.1). British Columbia (which implemented a carbon tax in 2008), Manitoba, and Ontario, also WCI members, plan to adopt trading programs in the future but have not specified dates (Western Climate Initiative 2012). Manitoba was also a member of the now-defunct MGGRA.

The resource-rich province of Alberta, which produces over 40 percent of Canada's GHG emissions, has adopted an approach based on reducing energy intensity. Large emitters, including oil producers, must reduce their emissions per unit of output by 12 percent against a varying baseline measurement between 2007 and 2010. If they cannot meet these targets, emitters can comply by one of three mechanisms: purchase excess credits from another emitter, purchase Alberta-based offsets, or pay C$15 per metric ton toward an energy/climate technology fund . Most allowances are allocated freely to emitters, and, as of 2010, the program covered 97 facilities that accounted for roughly 50 percent of Alberta's GHG emissions (Alberta Government 2011a, b, 2012). The program's modest intensity reduction targets means that even if emitters meet their targets, overall GHG emissions may well increase (for a critical discussion of this issue, see Doluweera et al. (2011)).

3.1.5 *New Zealand and Australia*

New Zealand launched an emissions trading program in 2008 which was designed to cover almost all New Zealand emissions with caps based on its 2008-2012 commitment under the Kyoto Protocol.[12] Since New Zealand is a small economy, the program was built around the idea of linking to other markets; this initially includes the Clean Development Mechanism but could be expanded to other national or regional carbon markets (such as the EU or Australia). This feature has made the program vulnerable to international policy uncertainty and to issues surrounding the CDM. The program covers a relatively small number of large emitters who must reduce emissions, purchase domestic or international offsets, or pay $25 (New Zealand) per ton of emissions. The program has no price floor and prices have steadily declined through 2012, generally following the movements of CDM prices. Industries facing international competition, horticulture, and fishing receive up to 90 percent free allocation, but the power sector, transportation, and forestry do not (New Zealand Government 2012a).

In Australia, after a long and contentious political process in which the prime minister reversed her public position, a series of bills became law in 2011 that will begin an emissions trading program in 2015. In the meantime, major carbon emitters will pay a steadily increasing carbon tax set by the legislation. Government revenues from this tax and auctions beginning in 2015 will go to new spending on efficiency, renewables, and technology, with at least 50 percent of revenues going to increased pension payments, increased tax credits, and decreased income

[12] New Zealand has not renewed its commitments under the 2013-2020 period of the Kyoto Protocol. However, its carbon market will continue to seek reductions based on negotiataions within the UNFCCC framework.

taxes for households (Australian Government 2012b). In August 2012, Australia announced that its program will link with the EU ETS, allowing emitters to surrender EUAs to comply with up to 50 percent of their requirements (Australian Government 2012a; Reklev 2012a). Additionally, European emitters may use Australian allowances for compliance as early as 2018. However, Australia's current opposition party has made repealing the carbon price "the top priority" on its agenda, calling into question the policy's viability moving forward (Australia Liberal Party 2012).

3.1.6 *Other Markets, Including Voluntary Markets*

Recent legislation passed in South Korea and Mexico has laid the groundwork for new national-level programs beginning in 2015. Much about these programs remains to be decided, but one interesting feature of Korea's program is that it will not allow international offsets for compliance until at least 2020 (Reklev 2012b). Additionally, some have speculated that Mexico's new president, Enrique Peña Nieto, is unlikely to implement much of the recently passed climate legislation (Teixeira 2012). China has established a series of local and regional pilot programs, while programs under discussion in India, Japan, Vietnam, and Thailand indicate an interest in cap and trade across much of Asia. Other emissions trading proposals are currently under discussion or development in Brazil and Chile, among others (Hood 2010).

Finally, voluntary carbon markets refer to a variety of organizations that allow individuals or businesses to purchase offsets from emissions reduction projects. Since 2002, voluntary markets have grown from $43 million in revenues to a peak of $705 million in 2008; they stood at $572 million as of 2011 (Ecosystem Marketplace and Bloomberg New Energy Finance 2008-2012). Dozens of organizations offer voluntary carbon offsets, and their standards for evaluating and monitoring GHG reduction projects are typically less stringent than those used for the CDM or JI. One benefit of less stringent standards is reduced bureaucracy and the potential for lower project costs; however, weaker standards could also lead to certification of projects that do not provide their stated benefits (Benessaiah 2012).

3.2 Key Design Choices

3.2.1 *Setting the Cap and Comparability Issues*

A fundamental question for any cap-and-trade program is the level of the cap. Jurisdictions have taken a variety of approaches to setting reduction targets and measuring progress. As noted in Table 1, some programs explicitly base cap levels on the Kyoto Protocol targets and 1990 base year, but many have developed alternative targets and base years. Absent a common approach to setting targets, sharing responsibility for emissions reductions across jurisdictions, or measuring progress, it is difficult to compare the goals of one program to the next without controversy.

Adding to the comparability challenge is the fact that most programs do not cover all economic activity. For example, a 21 percent reduction in the cap specified by the EU ETS does not represent a 21 percent GHG emissions reduction across the entire economy. Instead, the EU ETS is one of several tools implemented by the EU to reach its Kyoto Protocol targets. Australia and New Zealand, on the other hand, cover all or almost all economic activity with their trading systems and will attempt to reach their pledges under the 2011 Copenhagen Accord almost entirely through emissions trading. Most sub-national programs in the United States and Canada neither cover the entire economy (with the notable exception of California, which will do so) nor explicitly link their reduction goals to any international agreement.

Clearly, researchers face a variety of challenges in comparing regional, national, and sub-national programs, especially when those programs are designed to address a global problem. We explore this issue further in section 5.3.

3.2.2 *Regulated Sectors and Allocation*

Since greenhouse gas emissions occur as a by-product of virtually every form of economic activity, governments face the crucial questions of which sectors to regulate and whether to auction allowances or allocate them free of charge. After an early tendency to provide free allocations, new programs and the revised EU ETS have moved toward auctioning allowances (we discuss some reasons for this in section 4.4).

The EU ETS, RGGI, and other programs have focused on covering large point sources such as power plants and factories. Such sources have relatively low monitoring costs and—from a political perspective—are often easier to target for regulation than sources such as transportation. Additionally, most electric power markets do not face substantial competition from foreign firms, meaning that many power producers can pass along costs to consumers. As a consequence, electricity generators in most programs receive fewer free allowances than other types of emitters. The NZ ETS, for example, gives no free allowances to the power sector, and RGGI has given just 4 percent in its 2009-2011 period.

Heavy industrial processes (e.g., cement, aluminum, lime, and fuel refiners) are covered by every system other than RGGI, but, fearing economic dislocation, governments have given substantial free allocations to these firms. In every trading program that includes them, these typically trade-sensitive industrial sectors receive up to 95 percent free allocation. Regulators generally calculate a sector or firm's level of free allocation based on formulas including exposure to foreign competition and each sector's ability to reduce their emissions intensity. It is worth noting that a substantial share of these free allowances may be going to producers who do not require free allocation in order to continue operating domestically. As a result, these producers reap "windfall profits."

3.2.3 *Cost and Volatility Containment*

We begin the discussion of costs by noting that market-based emissions reduction programs typically offer substantial savings over traditional standards-based regulation (Stavins 1998; Carlson *et al.* 2000). Among market-based programs, normative theory emphasizes the value of either fixing or constraining prices, as noted in section 2.2. Yet the political debate and press coverage has put more focus on avoiding *high* prices. This may be unsurprising, given that a market-based program inevitably broadcasts prices much more transparently than standards-based regulation. Regardless of the motivation, significant uncertainty over abatement costs has led program designers to include mechanisms to prevent allowance prices from exceeding economically and politically tolerable levels.

The emissions trading programs discussed above have typically turned to one or more of the following three types of cost containment. First, regulators can impose a price ceiling, allowing emitters to purchase allowances directly from the government at the ceiling price. For example, participants in California or Quebec's program will be able to purchase credits for $40–$50/ metric ton, essentially capping trading prices (Western Climate Initiative 2012).

Second, emitters may be allowed to bank or borrow allowances from past or future compliance periods. This option allows emitters greater flexibility in meeting emissions reduction targets over time without forcing them to precisely achieve each interim target (as noted in section 2.1, this type of flexibility is justified on cost-effectiveness grounds alone). The EU ETS, which began with no banking or borrowing between phases, now allows unlimited banking of credits (Ellerman *et al.* 2010). Although it would also reduce compliance costs, borrowing provisions have been less widely adopted (Fell & Morgenstern 2009).

Third, high market prices can trigger provisions that relax the constraints of the program.[13] In RGGI, for example, if carbon prices reach $7/ metric ton, emitters are allowed to purchase more carbon offsets to meet their compliance needs. If prices reach $10/metric ton, emitters may purchase still more offsets to reach their targets (Regional Greenhouse Gas Initiative Inc. 2012).

Some programs also employ price floors to prevent market prices from falling below a certain level. Auction price floors are used in RGGI and California along with anticipated programs in Australia and Quebec. These floors are not implemented as an allowance buy-back, but instead set a price below which allowances will not enter the market. Price floors may seek to accomplish two goals: one, to maintain prices at a level where firms have an incentive to invest in emissions reduction technology, or two, to provide a steady source of revenue for governments.

3.2.4 *Offsets*

International emissions offsets offer a very large potential pool of mitigation opportunities, providing industrialized nations a lower-cost option for GHG reductions relative to reducing emissions within their own borders (Weyant & Hill 1999). Domestic or local offsets can also offer cost savings relative to opportunities within a given cap-and-trade program, but represent a smaller universe of activities compared to international offsets. Although specific provisions and restrictions vary, all programs to date employ offsets in some capacity.

Offsets from the CDM and JI play a major role in the EU ETS, with some nations allowing up to 20 percent of emissions reductions to be met with offsets. (Ellerman *et al.* 2010). Participants in the EU ETS may not purchase CDM or JI offsets from forestry or land-use change projects, due largely to questions over the permanence of carbon sequestration in forestry projects (Kim *et al.* 2008). Additionally, some have raised concern over potential harm to indigenous peoples due to forestry projects (Stickler *et al.* 2009).

While international offsets have played by far the largest role to date, regional or local offset programs to reduce emissions exist in several established and emerging trading programs. The notion that offsets should take place within a program's borders, instead of from

[13] Exactly how these latter mechanisms might affect behavior is sometimes unclear, particularly when specific actions trigger a significant and well-understood price decline. With banking, such an expected decline is at odds with a typical no-arbitrage condition.

international CDM or JI projects, has taken hold in some cases. Offset projects based within a program's borders ensures that the associated investment stays close to home, to the benefit of local economies. On the other hand, local offset projects may cost more than offsets provided abroad, implying higher costs for locally regulated industries.

In New Zealand's program, where no auctions are held, emitters comply with any unmet reduction goals by purchasing offsets, either from New Zealand-based forestry offsets or from CDM or JI projects (New Zealand Government 2012b; Point Carbon 2012b). Regional programs in North America, such as RGGI, California, and Quebec, have thusfar avoided the CDM, instead giving preference to offsets from regional or domestic emissions reduction projects. These North American programs allow offsets to make up just a small share of compliance, so firms do not need to draw on the millions of metric tons of offsets available from the CDM or JI. Perhaps more importantly, CDM and JI credits have only been available to parties with commitments under the Kyoto Protocol.

3.2.5 *Market Monitoring and Oversight*

After the 2008 financial crisis, virtually all financial markets came under new scrutiny. Carbon markets were no exception, and new proposals for trading programs in the United States came with calls for strong oversight. In fact, the 2010 Dodd-Frank financial reform and consumer protection bill created an interagency working group to conduct a study on maintaining and increasing transparency for carbon markets (Interagency Working Group for the Study on Oversight of Carbon Markets 2011). Similarly, an EU directive adopted in 2011 (the Markets in Financial Instruments Directive, or MiFID) will significantly expand oversight of carbon markets (European Commission 2012b). Primary goals for market oversight include facilitating price discovery, ensuring transparency and access to information, and preventing manipulation or abuse in the marketplace.

Monitoring and oversight occurs in the primary (first purchase or issuance), secondary, and derivatives markets. In primary markets, regulators typically seek to track initial ownership of allowances, the conduct of allowance auctions, and the creation and verification of offset credits. As an example, RGGI tracks allowance ownership through its CO_2 Allowance Tracking System (COATS), employs an independent monitor to review auctions, and requires member states to verify emissions offset programs (Regional Greenhouse Gas Initiative 2010).

Oversight of secondary markets, where spot transactions occur and futures contracts are created, presents a different set of challenges. Equal access to information is vital to ensure an unbiased marketplace, and programs seek to ensure this by announcing new market data at pre-determined intervals. Additionally, regulators may seek to limit market power of any single entity by limiting positions, as seen in California and Quebec's forthcoming programs .

Derivatives based on emissions allowances are typically subject to the same market oversight as other derivatives instruments. In the United States, oversight of these instruments and exchanges falls under the purview of the Commodity Futures Trading Commission (Interagency Working Group for the Study on Oversight of Carbon Markets 2011).

4. Market Experience and Lessons

Having explored the initial design features associated with different carbon markets, we now turn to market experience and evolution. We focus on the EU ETS, CDM, and RGGI, for which there are at least several years of data (see Figure 1), but we also include recent information on New Zealand and California, where futures contracts began trading in 2011. We draw a number of lessons based on the experience to date, highlighting price levels, market operation, banking, allocation and revenues, leakage, and offsets.

We do not focus directly on the Kyoto Protocol, as its primary compliance instrument, called Assigned Amount Units (AAUs), are typically traded in one-off transactions, negotiated and sold directly from one Annex-I nation to another (Aldrich & Koerner 2012). Although nations have traded millions of AAUs in these transactions, standardized exchanges have not emerged. Instead, the Kyoto Protocol's primary contribution to emissions trading markets has been the development of the EU ETS and the CDM, which we do explore below.[14]

Figures 2 and 3 provide basic information on carbon prices and volumes. Carbon prices in all markets have been falling since 2008 in response to the current economic recession. Nonetheless, volumes have been increasing, both in terms of activity within markets as well as the creation of new markets. The EU ETS has dominated the marketplace, with far greater volumes, liquidity, and price volatility than any other market. The EU ETS is also the largest outlet for CDM and JI credits. Additionally, the EU system is the only one where a significant secondary market has developed, with market participants buying and selling standardized contracts up to five years in advance on a variety of exchanges. While trading in the EU ETS began mostly with non-standardized over-the-counter (OTC) transactions, exchange-based trading likely surpassed OTC volumes sometime in 2008, indicating increased levels of standardization and liquidity (Ellerman *et al.* 2010).

4.1 Lesson: Positive Prices Imply Emissions Abatement, But How Much Is Unclear

The mere presence of a consistently positive price on carbon suggests that these trading programs are having at least some impact on behavior with regard to emissions levels. Program design, underlying fuel prices, and larger economic forces all impact behavior of market participants, but due primarily to the youth of the EU ETS and other programs, research on the extent of each of these impacts relative to the carbon price remains limited.

One way to approach the abatement question is to estimate emissions reductions based on elasticities derived from related analyses. A rough analysis of projections from the proposed U.S. Waxman-Markey legislation suggests emissions semi-elasticities of 0.0015–0.0061 in the year 2015.[15] That is, for each $10/ metric ton increase in the price of U.S. CO_2 allowances, emissions

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[14] Note that the EU ETS provides for corresponding adjustments in AAUs as EU allowances are transferred among EU countries. See EU Directive 2003/87/EC.

[15] Author's analysis of data from Energy Information Administration, "Energy Market and Economic Impacts of H.R. 2454, the American Clean Energy and Security Act of 2009."

during 2012–2015 would fall between 1.5 and 6 percent compared with a scenario with no price on CO_2 emissions. If similar economic dynamics are at play in the EU ETS, an allowance price of $16/metric ton (the Phase I average for the EU ETS) would suggest that the program resulted in reductions of 2–9 percent compared with BAU.

Existing research falls in line with this rough calculation. Empirical research on Phase I of the EU ETS suggests that during 2005–2007, emissions fell by 2–5 percent compared with BAU (Ellerman *et al.* 2010; Anderson & Di Maria 2011). If accurate, this level of abatement is consistent with the notion that Phase I of the EU ETS was less an ambitious carbon-reduction plan than an attempt to gain experience for future emissions reduction efforts (indeed, the Kyoto Protocol did not take effect until 2008). Since Phase II of the ETS has only recently concluded, we lack empirical research on abatement for 2008–2012. However, using the semi-elasticities noted above, the $20/metric ton average price of EUAs from 2008 through mid-year 2012 implies a reduction of 3–12 percent compared to a BAU scenario with no emissions trading system in phase I or II. With prices falling to one-third of that level at the beginning of 2013, the potential for emission reductions in phase III remain to be seen.

A key question—and sometime criticism—of current market-based policies is the degree to which they encourage long-term investment in new technologies rather than solely short-term fuel-switching and energy conservation. Much has been written about the importance of long-term investment to address climate change and the potential effectiveness of market-based policies to drive it (Jaffe 2002; Newell 2010). However, markets for CO_2 may be too new to fully inspire the long-term confidence to make those investments, and early research into the EU ETS suggests that such investments may be limited (Leiter *et al.* 2011). In the case of RGGI, emissions have fallen well below the cap due to fuel switching and weak economic growth, which likely limits investment in long-term emission reductions.

4.2 Lesson: Despite Some Rough Patches, Markets Have Generally Matured and Operated Effectively

Given the youth of the carbon markets, some learning and evolution is to be expected. Despite the exceptions noted below, markets have generally functioned well, enabling price discovery, increasing market transparency, and reflecting larger economic trends. Although smaller markets like RGGI (where the cap has been non-binding) and New Zealand do not see high volumes of trading, market depth and liquidity in the EU ETS is extensive.

One initial challenge in the EU ETS was the availability and quality of baseline emissions data. The EU ETS's pilot phase caps were constructed under time pressure and with a shortage of reliable data . When the first tranche of actual emissions data was released in 2006 by the European Commission, market participants were surprised by the low emissions levels vis-à-vis allowance supply, which sparked a dramatic fall in prices. This issue was also related to the inability to bank credits, which we discuss below in section 4.3. Large price swings directly related to data problems appear to be mostly absent since 2006–2007. However, governments have come under some criticism for not releasing timely and detailed data on individual allowance trades and holdings (de Perthius 2011).

A second issue affecting the EU ETS has been the types of offsets allowed under the CDM, which we discuss in section 4.6. Thirdly, the EU ETS has faced three high-profile cases of market manipulation, two of which were not specific to emissions trading markets. One of these two cases involved traders manipulating value-added tax (VAT) laws in different countries to defraud governments of over €1 billion from 2008 to 2009, while the second involved cyberattacks that likely stole over €50 million worth of allowances on spot exchanges in 2011 (de Perthius 2011; Frunza *et al.* 2011). The one major controversy unique to emission markets occurred when CDM credits previously collected by the Hungarian government for compliance re-entered the market. It appears that the Hungarian government simply swapped the CDM credits for another type of carbon asset under the Kyoto Protocol that it needed to sell. While the swap was completely legal under the Kyoto Protocol, it was surprising to many participants in the European Emissions Trading System and created the appearance of credit "recycling" that would have negated relevant carbon reductions and diminished the integrity of the trading system (de Perthius 2011).

After each instance of market manipulation, the European Commission revised its rules to address the specific problem. For the VAT fraud and CER recycling issues, the Commission has increasingly centralized oversight of the EU ETS, making it harder for individuals to take advantage of differences between national-level laws and regulations. The cyberattacks, which forced a multi-day suspension of spot trading, prompted the Commission to re-examine the security measures put in place by the various exchanges that facilitate trading (European Commission 2010).

As for RGGI, the program's independent market monitor has found no major irregularities since trading began in 2008 (Potomac Economics 2009, 2010, 2011). Market and auction data is released by RGGI regularly, and allowance holdings are traceable online through the program's CO_2 Allowance Tracking System (COATS).[16]

A final issue for market behavior has been uncertainty stemming from the central role of policy. Since emissions markets (excluding voluntary markets) are created by governments, changes in policy have the potential to dramatically impact those markets. As the EU ETS enters its third phase, the European Commission is reportedly considering a delay in auctioning millions of EUAs, which would likely drive up short-term prices (Allan 2012; Szabo 2012). However, unless those allowances are retired permanently, they eventually would come into the market, suppressing future prices. Additionally, program changes intended to reduce the supply of EUAs may decrease the level of certainty market participants can expect from the European Commission, potentially deterring investors from participating in the futures market or from banking allowances.

Carbon markets in the United States and Australia also face major uncertainty over policy. RGGI saw a decrease in the size of its market when New Jersey withdrew from the program in 2011. As mentioned earlier, Australia's opposition Liberal Party has vowed to eliminate that country's proposed emissions trading program if it gains control of the government in 2013 (Australia Liberal Party 2012). In New Zealand, rules were revised to allow one allowance to be surrendered for two metric tons of emissions during a transition phase (Fallow 2009). We return to this issue in section 5.2.

[16] See www.rggi-coats.org for details.

4.3 Lesson: Banking Matters

As noted above, the first release of actual EU ETS emissions data in 2006 showed emissions levels charting a course well below their Phase I cap. Because EUAs in Phase I could not be banked and could only be used between 2005 and 2007, an oversupply of permits meant that prices were likely to drop—and drop they did.[17] In the first quarter of 2006, spot EUAs traded at €25/ metric ton. By the final quarter of 2007, spot prices were essentially zero, at €0.06/metric ton (see figure 2).

This precipitous drop did not represent the underlying price of carbon, nor was it solely a problem of limited data; rather, it reflected the inability of market participants to bank allowances for use in future phases. Had emitters been able to use their remaining Phase I credits in Phase II of the EU ETS, prices would still have dropped in response to news of a current-period oversupply but certainly would not have approached the near-zero levels seen in late 2007. The rationale for not allowing banking was the desire to separate Phase II, which coincided with the first Kyoto compliance period, from the EU ETS' Phase I trial period—but the consequences of this decision were clear. Spot prices for Phase I allowances approached zero, even while contract futures prices for Phase II EUAs hovered above €20/metric ton (Point Carbon 2012a). The European Commission, as well as designers of future programs, took careful note. In current and future phases of the EU ETS, as well as all other major emissions trading programs, unlimited banking of permits between phases is permitted.

An emerging question is exactly how much banking an emissions trading system can (and should) support. Analyses of the failed Waxman-Markey bill suggested that by 2022 (10 years after trading's hypothetical start date), firms would have accumulated banked credits of 7–12 billion metric tons of allowances, 140 to 240 percent of the program's 2022 cap of roughly 5 billion metric tons (Energy Information Administration 2009). Recent EU ETS estimates suggest compliance entities, banks, and other market participants are banking nearly 2.5 billion allowances, roughly 119 percent of Phase II's annual cap for carryover into Phase III (Neuhoff *et al.* 2012).

From a normative perspective, we would want market participants to be as well informed as possible about the direction of future policy and to bank accordingly. However, if policymakers behave in the unexpected ways noted at the end of section of 4.2, this may have consequences. For example, New Zealand's late decision to allow one allowance to count for two metric tons of emissions effectively halves compliance prices. The reverse would be true if the EU chooses to hold back auctioned allowances. Even if temporary, these experiences of having the government step into the market will influence the way market participants view the rigidity of the emissions cap and consequent allowance prices. An important, unanswered question is the degree to which governments ought to be disciplined or flexible in the face of various events.

[17] While 2005–2007 EUAs could not be banked, CDM credits could be moved forward. However, few CDM credits had to this point in the program been issued.

4.4 Lesson: Allowance Allocation Can Involve Large Revenues and Distributional Impacts

Every major emissions trading program allocates at least some free allowances to heavy industry and, in some cases, power generators. In Phase I of the EU ETS, most NAPs compelled power generators to reduce their emissions more than other sectors. Concerned about a consequent rise in consumer electricity prices and encouraged by industry, regulators and elected officials allocated the power sector a large share of allowances at no charge (Markussen & Svendsen 2005). In competitive markets such as Germany, power generators passed along the opportunity costs of these free allowances to their customers, allowing generators to extract rents roughly comparable to their proportion of freely allocated allowances (Sijm *et al.* 2008; Ellerman *et al.* 2010). Put simply, power companies effectively charged customers for permits they received for free.

This predictable market outcome demonstrated that free allocation did not necessarily protect consumers but did have distributional consequences, and was not warmly received by some observers (Gow 2006; Harrison 2009). The European Commission has responded by revising the program's guidelines, limiting free allocations and increasing the proportion of allowances sold at auction. Because heavy industry faces significant international competition, which tends to prevent firms from passing on allowance or abatement costs, heavy industry firms will continue to receive substantial free allocations. In contrast, power generators will receive limited free allowances, and only in specified nations.

Despite well-known public finance arguments against earmarking, governments may also seek to put rents extracted from emissions trading programs to particular purposes. In many programs (such as in Alberta, Australia, California, and RGGI), governments must put a share of revenues raised from auctioning emissions toward low-carbon energy projects. These investments include renewable energy projects, carbon capture and storage technology, and efficiency programs. RGGI states, for example, invested over 60 percent of their auction revenues (roughly $480 million) toward energy efficiency and renewable energy programs between 2008 and 2010. Other revenues from RGGI went toward direct energy assistance to households, and a small share went to non-related government projects (Regional Greenhouse Gas Initiative Inc. 2011).

4.5 Lesson: Significant Competitiveness Impacts and Emissions Leakage Are Not Inevitable

In the absence of a global regime, reduced economic competitiveness of covered sectors is often mentioned as a key concern with carbon taxes or emissions trading programs. Two distinct issues arise in this context. First, an emissions trading program entails economic costs and can redistribute economic activity—both jobs and capital—away from regulated sectors, particularly emission-intensive manufacturing. These costs and redistribution are economic issues. In addition, some of this manufacturing activity may shift to unregulated jurisdictions, with domestic emissions reductions reappearing elsewhere in the world. In addition to amplifying the economic issues, such emissions leakage is an environmental issue. If significant, leakage draws into question the underlying environmental rationale for emissions trading programs. However, a review of the (limited) empirical literature indicates that, at least for the early phases of the EU ETS and RGGI, competitive losses and leakage appear to have been small in the few sectors where it has occurred.

Ellerman et al (2010) found "no observed impact" on competitiveness in the oil refining, cement, aluminum, or steel sectors during Phase I of the EU ETS. Demailly and Quirion (2008) found that Phase I of the EU ETS created only a small loss of competitiveness in the iron and steel sectors. Lacombe (2008) found a similar limited impact on the EU refining sector during Phase I. An analysis of the EU aluminum sector by Reinaud (2008) found no statistical evidence of negative competitiveness impacts from the program but notes that information gaps remain.

However, one survey of firm managers suggests larger competitive impacts and associated leakage in certain sectors. For example, 55 percent of survey respondents in metals manufacturers, and 44 percent of pulp and paper and cement, lime, and glass manufacturers, stated they have either moved or are considering moving out of the EU ETS compliance zone, while only 14 percent of other firms stated they have moved or are considering such a move (Point Carbon 2011). There are reasons to regard such survey results with caution. First, the survey consists of a fairly small sample size (215 respondents for the entire EU economy). Second, respondents may not disaggregate the cost of CO_2 from the overall cost of electricity, potentially leading to an overemphasis on the weight of carbon prices.

If the lower estimates from the EU are accurate, these minimal competitiveness impacts and leakage rates may reflect the modest GHG reduction targets implemented in the first phase of the EU ETS. Additionally, uncertainty over emissions reduction policies around the world may delay firms from making decisions on moving to jurisdictions where GHG policies could enter into effect after the firms' relocation.

These observed competitiveness impacts and leakage generally fall below the levels predicted by some analyses of emissions pricing in the United States. Aldy and Pizer (2008) estimated that a $15/metric ton carbon price in 2012 would result in decreased heavy industrial production by 1.6–3.4 percent. Ho, Morgenstern, and Shi (2008) estimated that a $10/metric ton price would decrease production of most trade-exposed heavy industry by between 1 and 3 percent, though some industries would see decreases up to 7.7 percent. A U.S. interagency analysis of the proposed Waxman-Markey bill (Interagency Competitiveness Analysis Team 2009) estimated that the legislation would result in an increase in marginal production costs of between 0 and 4 percent. Finally, Fischer and Fox (2009) estimated that a high ($50/metric ton) carbon price could result in leakage rates of up to 27 percent for some energy-intensive sectors.

Leakage and competitiveness issues in RGGI have also been a concern. Since RGGI states operate in an electricity market integrated with non-RGGI states, the potential for leakage in the power sector clearly exists. Some research has suggested leakage rates ranging from 28 percent with $3/metric ton prices to 90 percent with $7/metric ton prices (Chen 2009; Wing & Kolodziej 2009; Kindle *et al.* 2011).

However, low carbon prices resulting from a weak economy and historically low natural gas prices may have prevented extensive leakage in RGGI. One 2008–2010 empirical study in New York State suggests that allowance prices were too low to cause leakage (Kindle *et al.* 2011).

4.6 Lesson: Offsets Can Work, But They Are Complex

The Kyoto-established CDM and JI mechanisms have supported tens of thousands of emissions reduction projects in the developing world, allowing the EU ETS and other trading programs to reduce costs while supporting mitigation efforts in developing countries. We will focus here on the CDM, as CDM projects have outnumbered JI projects by over 12 to 1 from 2007 to 2011 (Fenhann 2012). Exactly which types of projects should receive offset credits has received the most attention. For offsets to truly reduce emissions, credits can only be given to projects that would not have occurred without the offset credit program. That is, projects must provide additional emissions reductions compared with a world where the CDM did not exist.

Although much research indicates that the CDM has resulted in real emissions reductions, a variety of researchers have described cases where non-additional projects received credits under the program (Wara 2008; Elsworth & Worthington 2010; Lambert 2011; Zhang & Wang 2011). The most problematic project type in the past has been HFC-23, a refrigerant used in industrial processes that has roughly 10,000 times the global warming potential of carbon dioxide (United Nations Framework Convention on Climate Change 2012a). Because of its high global warming potential, projects that reduce HFC-23 receive large amounts of credits.

Figure 4 shows how, despite a very small proportion of projects, HFC-23 reduction projects received a large share of reduction credits from 2005 to 2006. Lambert (2011) finds that inadequate baseline measurements helped create a perverse incentive, encouraging HFC-23 emitters to temporarily *increase* their emissions, allowing them to later reduce HFC-23 output and claim thousands of valuable credits. In the wake of such research, the European Commission voted in 2011 to disallow credits from projects reducing HFC-23 (European Commission 2011). Additionally, the CDM Executive Board revised its guidelines regarding HFC-23 and other similar gases, leading to an increased number of rejected project proposals.[18] Figure 4 shows the result: HFC-23 projects have received far fewer offset credits since 2006.

The number of proposed and implemented CDM projects has grown substantially, led by renewable energy such as wind, solar, or biomass. Still, overall issuance of CDM credits has decreased steadily since 2007. This trend likely reflects stricter rules established by the CDM Executive Board, as well as the fact that many planned renewable projects have not yet begun receiving offset credits.

In Phase II of the EU ETS, the usage of offsets varied considerably among EU nations, ranging from a low of 0 percent in Malta and Liechtenstein to 13 and 18 percent in Spain and Lithuania respectively. Overall, the use of CERs for compliance across all EU nations from 2008 through mid-2012 was roughly 6 percent, totaling 456 million CERs. When ERUs earned from JI projects are included, the figures reach over 7 percent, or roughly 555 million international offsets.[19] However, Phase II NAPs allowed EU nations to use offsets for an average of 11 percent of their emissions reductions, meaning that emitters have chosen not to use their

[18] Author's analysis of CDM data from Fenhann (2012). "CDM projects," accessed via www.cdmpipeline.org.

[19] Author's analysis of data from European Environment Agency, http://www.eea.europa.eu/data-and-maps/data/data-viewers/emissions-trading-viewer.

maximum number of offsets. Germany and the United Kingdom, the region's two largest GHG emitters, have used fewer than half of their allowed offsets for compliance through Phase II.[20]

This limited use of offsets in Phase II is likely related to the ability of firms to bank their ability to use offsets into Phase III of the EU ETS. If firms expect allowance prices to rise (or credit prices to fall), they may refrain from using offsets in the current period.

The CDM has also presented a variety of distributional questions due to the associated transfer of resources from Annex-I nations to developing nations. Through 2011, by far the largest share of projects and credits were going to China and India. In fact, between 2006 and 2011, over half of each year's CDM credits went to projects in China (topping out at 75 percent in 2007) (Fenhann 2012). Since China is the world's largest carbon emitter, it is not surprising that a large share of GHG reduction projects would flow there. However, nations or political stakeholders that believe China should commit to a more stringent emissions reduction plan, or see China as a competitor, may object to the transfers enabled by the CDM .

Most researchers agree that the CDM has successfully produced real emissions reductions, but whether the projects are meeting broader development objectives, such as economic growth or technology transfer, remains uncertain (Lecocq & Ambrosi 2007; Olsen 2007; Sutter & Parreno 2007; Dechezlepretre *et al.* 2008; Schroeder 2009; Popp 2011).

In light of the concerns described above, and coupled with rapidly falling prices (see figure 2), rules for CDM projects continue to evolve. Currently, a variety of CDM project types face review from the CDM Executive Board and the EU ETS, where over 7,900 CDM or JI projects have been approved, and nearly 3 billion offset credits have been allocated (CDM & JI Monitor 2012; Fenhann 2012). Meanwhile, members of the European parliament have called for restrictions on CDM credits for certain types of projects (McGarrity 2012).

It is impossible to conclude a discussion of offsets without at least noting the collapse of the CDM market at the end of 2012. After remaining around €10-15 range for most of 2009, 2010, and the first half of 2011, CDM prices fell steadily to less than €1 in November and December of 2012. This has been ascribed to increased limitations on the use of CDM credits in the European Union, uncertainty about future demand, and increasingly robust supply – issues that will need to be sorted out for investors to continue to have confidence in offset markets.

5. The Future of Carbon Markets: New Issues

We are now at a very different place than we were fifteen years ago. In the late 1990s, most intellectual and stakeholder debate focused on a single global trading program being designed as *the* vehicle to address global climate change.[21] The key issues were the design of that program. Today, that form of top-down global program seems far away, if not impossible.

[20] Percentage of offsets allowed through NAPs from Ellerman et al (2010). This may reflect lingering benefits of the UK shift from coal to natural gas and German re-unification, both of which led to significant emission reductions against 1990 baselines.

[21] Hahn and Stavins (1999), for example, consider how an international trading system will interact with domestic policies that may or may not involve emissions trading.

Instead, we see a multiplicity of national and even sub-national trading programs emerging. Moreover, we now have real experience with these carbon markets. At the same time, the future of national climate change policy in the United States—the largest developed country emitter and original protagonist of emissions trading—is uncertain. If a comprehensive U.S. policy does emerge, it is not clear whether that policy will be in the form of an emissions trading program, an emissions tax, a tradable performance standard, or traditional regulation. This raises a number of new issues that have received little attention in the previous literature and that in some cases were not fully anticipated or understood during the design stages of existing carbon market systems.

5.1 Linking Carbon Markets

Front and center in the new discussion of carbon markets is how, whether, and when different markets can be linked so that regulated entities in one jurisdiction can use allowances or credits from another jurisdiction for compliance, and possibly vice-versa (Jaffe *et al.* 2009). There are a variety of motivations for these kinds of linkages: achieving global cost savings and gains to trade, reducing domestic market volatility, lowering domestic compliance costs, creating momentum for global action, addressing potential concerns of major trading partners, and creating the framework to become a net exporter of emissions credits, among others. At the same time, there are challenges to linking, most notably the risk to environmental integrity, the need to harmonize features (and corresponding loss of sovereignty over program design), and the distributional consequences of higher (or lower) prices.

Much of the linking work has focused on the mutual gains to trade. Indeed, the early analysis of the Kyoto Protocol focused on how much cheaper a global trading system would be compared to a system of autarkic systems (Weyant & Hill 1999). As domestic emissions trading proposals began incorporating features like price caps, additional work began to show that certain features in one system could lead to increased emissions if two systems were linked. For example, Fischer (2003) shows that linking a system that is indexed to output with an ordinary capped system almost always increases emissions. As linking discussions have become more serious, people have begun to think about exactly which features have to be aligned to avoid such issues and which do not (Mace *et al.* 2008).

In practice, linkages may be one-way or two-way (Mehling & Haites 2011). In a one-way linkage, credits in one system can be used for compliance in another, but not vice-versa. In a two-way linkage, both systems mutually allow the other's credits to be used for compliance.[22] It is useful to further distinguish within each direction of linkage between the buy- and sell-linkage decisions and how they relate to aforementioned concerns (recognizing that any buy-linkage decision by one actor represents a sell-linkage decision, explicit or implicit, by the seller). A buy-linkage represents the decision by one trading system to accept for compliance allowances or credits created and offered for sale by another system. Sell linkages represent an implicit or

[22] Linkages can also be indirect: if A links to B and B links to C, A will have an indirect linkage with C. For example, A's credits can be used for compliance in B, freeing up B's credits to move into C. The net result would be credits leaving A and entering C.

explicit decision by one jurisdiction to allow or encourage other jurisdictions to use its allowances or credits for compliance.

Concerns about environmental integrity typically arise in the buying system (Mace *et al.* 2008). The buying system is the one deciding that the seller's credits or allowances are valid for compliance in the buying system (the selling system has already decided to use their own credits for compliance). Concerns about harmonizing features can arise in either system, depending on who has more power in the linking negotiation; this is frequently a function of the relative market size. Currently, for example, the EU set the terms for Norway, Iceland, and Liechtenstein to enter the EU ETS, and California's provisions have influenced the design of Quebec's market, which is expected to link with California. Meanwhile, the CDM as a seller of credits remains independent, although the EU has set certain conditions on the kinds of CDM credits it will accept.

Finally, distributional concerns tend to arise in the selling system. For the buying system, linking lowers prices with the same environmental outcome—something many programs desire. Lower prices generally mean less redistribution among various market agents. For the selling system, linking raises prices. There are still gains to trade for the selling system as a whole, but higher prices generally mean more redistribution among buyers and sellers within the selling system. For this reason, Australia initially planned to restrict international sales of its allowances—despite the net gains from trade—though it now plans to sell allowances into the EU market in 2018 (Jotzo & Betz 2011).

One of the more interesting (and unanswered) questions is how delinking might work. Recent events in Europe, with Greece contemplating an exit from the Eurozone, should certainly make stakeholders think twice about what might happen down the road. So long as linked trading systems maintain distinct units of account, however, the better analogy is a pegged currency system rather than a currency union. In that case, it would appear to be not too problematic should delinking be necessary.

5.2 New Information and Revision

The discussion of linking and possibly delinking, as well as the carbon market experience discussed in the previous section, highlights that policies very clearly will be revised and even overhauled as time passes. The EU's reported consideration of holding back allowances in Phase III and the New Zealand decision to allow one allowance for every two metric tons of emissions during 2010–2012 are prime examples. Unlike conventional regulation, where the financial consequences of policy revisions are limited to impacts on the value of physical investment, revisions to carbon market rules (or carbon taxes, for that matter) affect the value of financial liabilities. They also affect the value of financial assets – government receipts or, in the case of free allocation, the value of future free allocations. Banking provisions in most carbon trading programs lead all of these effects to occur much sooner: While policy changes may not take effect for several years, any future change to allowance prices will immediately affect current prices. An expected price decline in the future means current allowances are less valuable to hold while an expected price increase means the reverse.[23]

While desirable from a dynamic efficiency point of view, such changes in market value imply financial impacts and consequences regardless of whether the contemplated change happens. If the holders of the allowances tend to be exactly the same people who face compliance obligations, the net effect of changes in market value could be relatively small, as the market value of allowances will fluctuate along with the cost of their future compliance obligation. However, exactly how allowances are valued on balance sheets can create problems even for these businesses.[24] Of course, if individuals without obligations are holding allowances, the consequences could be quite dramatic. It is unclear whether such impacts on speculative holdings should be viewed as fair or unfair, but there are clear impacts.

What do these market impacts imply for the broader issue of policy revision? First, policy revision still must happen. One of the defining characteristics of climate change is uncertainty about impacts. Coupled with the interdependence of policies in different jurisdictions, revisions to carbon market policies are essential to long-term efficiency (Murray *et al.* 2009). While markets and affected stakeholders may crave certainty, governments cannot guarantee certainty where it does not fundamentally exist.

Second, governments should strive for transparent and orderly policy revisions. Many government agencies, from central banks, to regulators, to courts, make decisions on a regular basis that have significant market consequences. Regulatory agencies, courts, and legislatures all need to pursue market-sensitive decisions in a way that allows all market participants equal access to information as well as advance notice of the sequence and timing of the decision process. This is one area where the EU ETS has recently been criticized (Allan 2012; Szabo 2012).

Finally, and most relevant to the topic of policy design, there may be ways that emissions trading policies can be made more amenable to necessary revisions. For example, one legislative proposal in the United States would have implemented a very specific schedule for periodic five-year reviews and revisions, with presidential submission of recommendations shortly after the compliance year ends and expedited Congressional action within six months.[25]

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[23] This is a significant difference between pollution taxes and emissions trading. Changes in future tax rates affect future liabilities while changes in future emissions trading rules, through price arbitrage, can affect the current price and hence immediate liabilities. Both can affect current asset values and both can be fueled by speculation. However, this distinction of price arbitrage in carbon markets, but not with carbon taxes, implies that jurisdictions have more latitude to attenuate the financial consequences of future policy changes by using carbon taxes and announcing changes well in advance.

[24] For example, under the Dodd-Frank Act, certain types of transactions have capital and margin requirements, though it is unclear whether these requirements would apply to a carbon market were it to emege (Epifani *et al.* 2012).

[25] Low Carbon Economy Act of 2007, S. 1766, 110th Congress. See §102(b), §501(b), and §501(c).

Another option is to put these decisions into the hands of an oversight entity, similar to a central bank.[26] Such an entity would be responsible for periodic reviews and changes to the emissions limit or other rules and would have the flexibility to do so deliberatively and outside the explicitly political sphere of legislatures. Given experiences with central banks and monetary policy, this approach has some appeal. However, it would be more challenging to take this approach with climate change because of the continuing divergence of views about the appropriate level of response, even among experts.

5.3 Alternative Policies and Comparability

The halting history on emissions trading in the United States highlights the fact that we are heading into a world not only of decentralized, bottom-up emissions trading regimes with varying rules but one in which some jurisdictions may set aside emissions trading altogether and pursue emissions taxes or more traditional regulation. In late 2012, for example, there was an unusual confluence of interest in carbon taxes to address climate change and the burgeoning U.S. deficit (Chemnick 2012). South Africa has also shown interest in a carbon tax (RSA 2010). Meanwhile, policy-related emissions reductions in the United States over the past few years (and likely in the near term) have arisen from tighter regulations on automobile fuel economy and tailpipe GHG emissions, renewable electricity capacity additions associated with federal and state subsidies and mandates, and new power plant emissions regulations under the Environmental Protection Agency.[27]

This diversity of policy approaches was not altogether unexpected. Under the Kyoto Protocol, there is no requirement to implement a domestic emissions trading program. The EU ETS, which sits at the apex of the carbon market discussion, only covers roughly half of European emissions, with traditional regulation or taxes used elsewhere (a number of European nations also levy a tax on GHG emissions). And in the United States, where concerns about competitiveness dominated the emissions trading debate, the Waxman-Markey bill did not seek to require trading partners to have an emissions trading program, only a "nationally enforceable and economy-wide greenhouse gas emissions reduction commitment for that country that is at least as stringent as that of the United States."[28]

What this immediately raises, however, is the need to measure the comparability of policies. Comparability is important for carbon markets because, among trading programs, comparability is necessary for jurisdictions to consider linking. In addition, among the broader suite of policies, comparability is necessary to avoid escalating concerns over competitiveness and emissions leakage that could threaten the sustainability of emissions trading. The Kyoto Protocol solved this issue by having countries negotiate agreed targets for one another based on a 1990 baseline. During the lead-up to the Copenhagen Accord, there was considerable debate

[26] See discussion in Pizer and Tatsutani (2008) and Newell et al (2005).

[27] The economic downturn and low natural gas prices have had a further downward impact on emissions over the last few years.

[28] American Clean Energy and Security Act of 2009, H.R. 2454 as passed by the House of Representatives, §767(c)(1).

over this with regard to various countries' pledges.[29] Most discussions look at emissions reduction efforts in one of five ways: (1) emissions reductions versus a historic baseline (percent reductions compared to 1990, 2005, etc.); (2) emissions reductions versus a business as usual baseline (e.g., percent reductions compared to forecast levels in 2020); (3) reductions in emissions intensity (percent reductions in emissions per unit of gross domestic product, energy use, or power generation against a historical baseline or future projection); (4) reductions in emissions per capita; or (5) the realized carbon price. However, there is no agreement on which metric is best. Many raise practical issues (e.g., conversion among currencies or calculation of business-as-usual forecasts), and different metrics yield dramatically different views.

This question of comparability is only compounded when evaluating actual implementation of policies and their outcomes as opposed to economy-wide emissions pledges. That conversation is only just beginning as countries embark on negotiations over a new climate change agreement, one based on the 2011 Durban Platform for Enhanced Action, and as they grapple with domestic stakeholders who are frequently concerned about whether other major emitters, particularly their economic competitors, are undertaking their fair share. Unlike the situation in Kyoto, where the dimension of comparability was a relatively narrow range of deviations from 1990 emissions levels, future comparability discussions will undoubtedly be much more complex.[30]

5.4 International Negotiations

What does this imply for future international negotiations concerning carbon markets? Unlike earlier, Kyoto-style negotiations that focused on a sequence of top-down, larger-to-smaller emissions trading issues—national emissions caps, trading rules, and further details such as the CDM—the new, post-Durban negotiations will necessarily focus on the tools for a bottom-up approach. On the one hand, a new agreement will need to support concerns over comparability and transparency of effort. Those countries pursuing and already engaged in carbon markets will want assurances that other jurisdictions will do their fair share. This is important if support in those countries for carbon markets is to continue, and such assurances will guide future responses to competitiveness concerns.

On the other hand, a new agreement will need to focus on ways to provide institutional support for markets themselves. Aside from spurring the creation of the EU ETS and other trading programs, the CDM is arguably one of the most important contributions coming out of the Kyoto Protocol. Despite the various challenges it continues to face, the CDM has an institutional structure that allows it to evolve and benefits significantly from the credibility afforded to it by its role in the agreed international architecture. The negotiations could look for

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[29] See, for example, Levin and Bradley (2009); Pew (2011); and Jotzo (2010).

[30] Targets under the Kyoto Protocol ranged from 8 percent below 1990 emissions levels for the European Union to 10 percent above 1990 emissions levels for Iceland (http://unfccc.int/kyoto_protocol/items/3145.php). The targets were somewhat renegotiated in the Marrakech Accords, which established limitations on the use of forestry sinks for compliance with the original targets (http://unfccc.int/methods_and_science/lulucf/items/3063.php). This relatively narrow range of targets had dramatically different consequences for countries facing different growth rates or other structural changes after 1990.

ways that a wider variety of country contributions can be supported. For example, some developing countries may want something like "model rules" for establishing a domestic trading program that would presumptively link to developed country programs already utilizing the Clean Development Mechanism. While rules for carbon markets and other abatement programs can and may emerge organically without an anchor in international agreements, creating model rules could be valuable, particularly for the many countries that will be too small to pursue an entirely customized approach. There are also questions about the future of the CDM itself. Decisions in December 2012 will limit future access to the CDM to countries participating in the next phase (2013-2020) of the Kyoto Protocol. This approach steers the CDM away from a role in a decentralized global carbon market by limiting its relevance to the subset of Kyoto participants. To achieve efficiency, future negotiations should be creating opportunities for linkages, not blocking them.

6. Summary

The overarching messages from this survey are that carbon markets are sizable, they have been expanding rather than contracting, market rules are evolving and not static, and a new architecture based on a more organic, bottom-up international design raises a suite of new questions. A global emissions trading system has not emerged as the linchpin policy solution for climate change in the way it may have been envisioned in the mid- to late-1990s. Other policy tools—carbon taxes and traditional regulation—are being implemented or considered in a variety of sectors and jurisdictions.

The design of carbon markets is benefiting from experience. Experience with windfall profits from free allowance allocation has led to an increased use of auctions. Jurisdictions are learning to handle market-sensitive information in a more transparent and orderly manner, but there is still progress to be made. Efforts to moderate high and low prices are providing lessons on what works and what does not while also making the simple point that prices matter. Perhaps most importantly, we are seeing that carbon allowance trading can support emissions reductions and send market signals for future investment. However, the strength of those signals for future investment hinge on confidence in the emissions market, the underlying regulatory framework and its stringency, and the broader investment climate.

The evolving nature of carbon markets and associated design changes imply that confidence in the market cannot be one hundred percent. Governments cannot provide certainty where it does not fundamentally exist. Looking forward, however, authorities need to be clearer and more orderly about policy revisions and recognize the consequent impacts on market price, market participants, and future market confidence.

Among the many issues facing markets in the future, the emergence of multiple emissions trading programs has put front and center the question of how, whether, and when these programs will be linked together. While a variety of motivations drive interest in linking, and there are a variety of ways to create links, three key concerns have limited linking so far. Buyers tend to be concerned about environmental integrity, as the buying system is establishing that purchased allowances are valid for compliance in their system. The necessary harmonization of certain design features also means that one or the other system is giving up some sovereign control. Perhaps most importantly for sellers, selling systems will generally see increases in allowance prices, with potentially adverse distributional consequences (despite overall gains from trade). For this reason, linkages among trading systems have proceeded relatively slowly.

What is the role of international negotiations regarding carbon markets in this emerging, bottom-up world? One role is to address the issue of comparability among different trading systems as well as among emissions trading, taxes, and traditional regulation. Comparability among trading systems supports linking, and comparability more generally can help avoid escalating competitiveness concerns. The latter concerns, related to both emissions and economic leakage, represent one of the greatest challenges to the long-term sustainability of carbon markets. In addition to comparability, it will be useful to explore how international institutions can more directly support carbon markets in a more decentralized regime. The CDM is a significant contribution; perhaps there are others.

Fifteen years after the signing of the Kyoto Protocol and the creation of the first vehicle for carbon markets, the Kyoto model of a top-down, global carbon trading system is essentially over. Carbon markets are not, however, over. The challenge now is to figure out how they can work in a much more complex—but clearly more realistic—world.

Tables and Figures

Table 1. Selected Emissions Trading Program Provisions

	~Size (metric metric tons)	Basis for cap level	Covered sectors	Allowance allocation	Cost containment	Offsets
Australia	0M	5 percent below 2000 levels by 2020. Based on Kyoto targets.	Power sector, most heavy industry, transport, waste. Agriculture and deforestation are exempt.	Auctions begin in 2015. A$23/metric ton tax in 2012. In 2015, 66–95 percent free allocation to heavy industry.	No price floor. Ceiling set at A$20/metric ton above expected EUA price.	Up to 12.5 percent CDM/JI in 2015. Up to 100 domestic offsets after 2015; 50 percent may be CDM/JI. No offsets 2012–2015.
California	160M in 2013 400M in 2015	15 percent below 2012 levels by 2020.	2013: power sector, large industrial. 2015: fuel distributors (transport, nat. gas).	~2/3 of permits auctioned in 2013. Some free allowances to electricity, industry decreasing over time.	Auction floor of $10/metric ton. Effective ceiling of $40–50/metric ton in 2012.	Restricted use of CDM/JI. Four types of domestic offsets OK. Up to 8 percent offsets.
EU Phase I (2005–2007)	2.2B	National Allocation Plans.	Power sector, most heavy industry.	10 percent maximum auctioned. Up to 100 percent free allocation and power sector and industry.	No price ceiling/floor.	CDM/JI OK. 0–20 percent offsets allowed. No forestry or nuclear project credits.
EU design changes	2.1B	EU burden-sharing (II) and linear decline (III).	New industrial activity added in 2008; more added in 2013. Aviation added in 2012.	~50 percent auctioned in 2013. Free allocations mostly phased out by 2020.	No change.	Roughly 6–7 percent offsets OK through 2020.
New Zealand	8.2M	No explicit cap. Based on Kyoto targets.	All sectors of the economy phased in by 2015.	No auctions. 60–90 percent free allocation to industry based on energy intensity.	No price floor. Effective $25/metric ton ceiling in 2012.	CDM/JI OK, but mostly domestic offsets to this point. All reduction goals met with offsets.
Quebec	5M	15 percent below 2005 levels by 2020.	2013: Power sector, large industrial. 2015: Fuel distributors (transport, natural gas).	~10 percent auctioned in 2012, up to 50 percent by 2013. Some free allowances to electricity, industry decreasing over time.	Auction floor of $10/metric ton. Effective ceiling of $40–50/metric ton in 2012.	No CDM/JI in first phase; could be introduced later. Up to 8 percent of entity's allocation may be offset.
RGGI	150M	10 percent below 2009 levels by 2018.	Power sector only.	No free allocations.	Price floor at $1.93/metric ton in 2012. $7–10/metric ton price ceiling triggers additional offset allowances.	No CDM/JI. 3.3 percent maximum offsets in 2012. 5–10 percent maximum if price ceiling is reached.
Waxman-Markey (H.R. 2454)*	2.3B	17 percent below 2005 levels by 2020, 83 percent below 2005 by 2050.	Power sector, large fuel processors. Large industry by 2014.	~27 percent auctioned in first year. Free allocations to power sector, trade-exposed industry, and others based on energy intensity.	$10/metric ton floor. Effective $28/metric ton ceiling for first year.	30 percent max offsets per entity. CDM/JI OK unless program administrator deems otherwise.

Note: *Waxman-Markey bill (H.R. 2454) as passed by the U.S. House of Representatives in 2009.

Figure 1. Timeline for Selected GHG Emissions Trading Programs

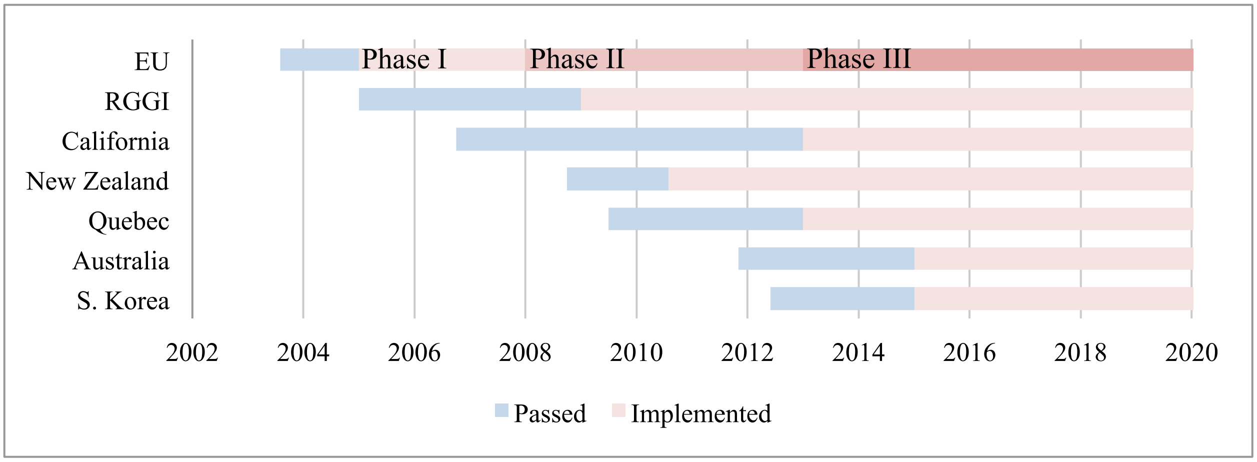
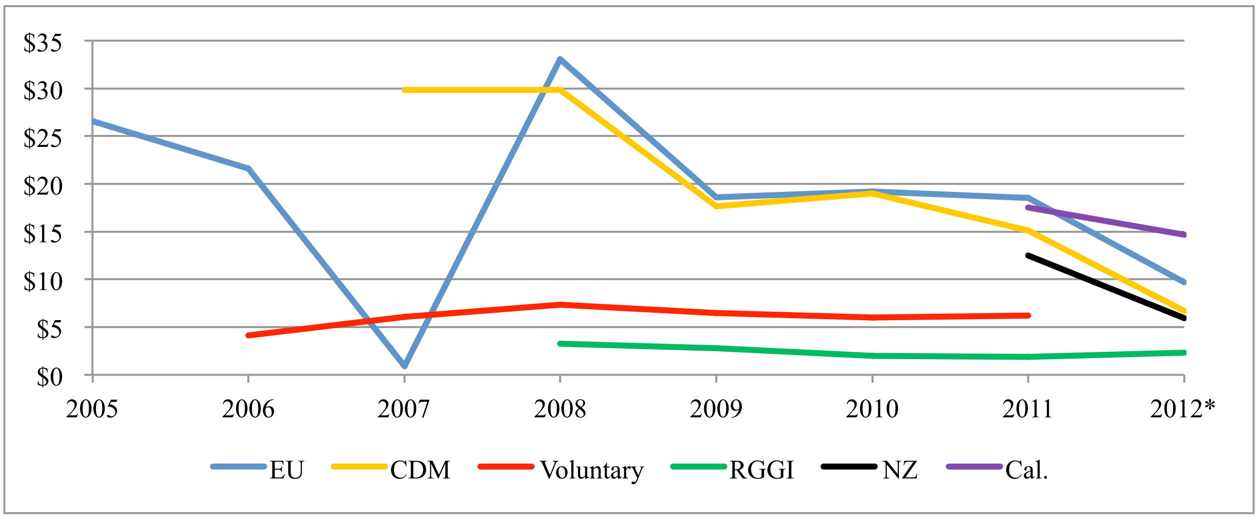
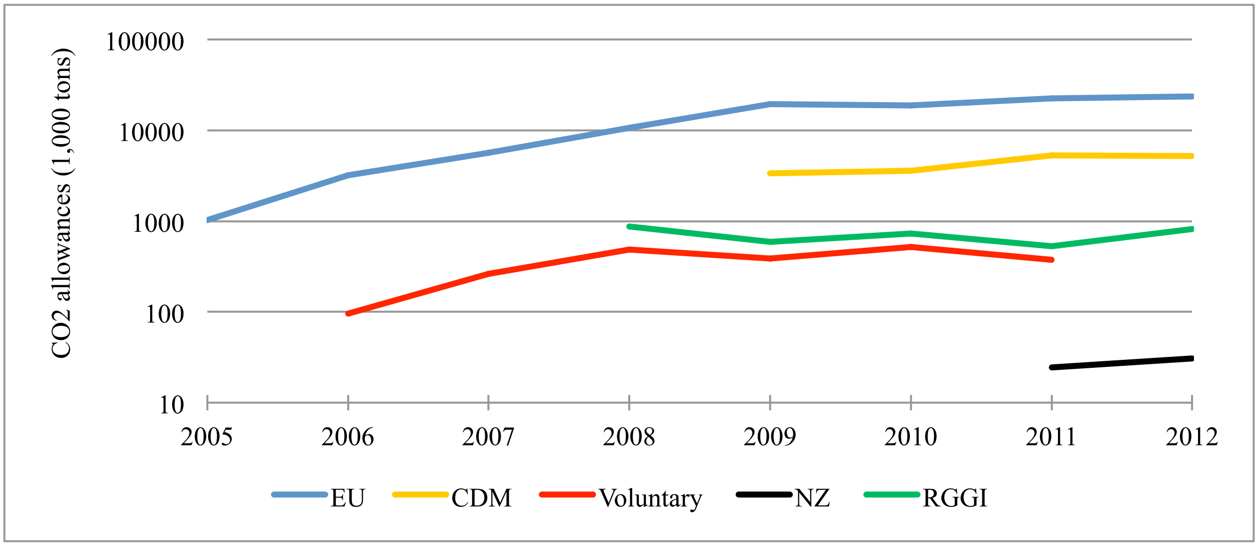


Figure 2. CO_2 Prices (Annual Average Price Per Metric ton CO_2, Nominal US$)



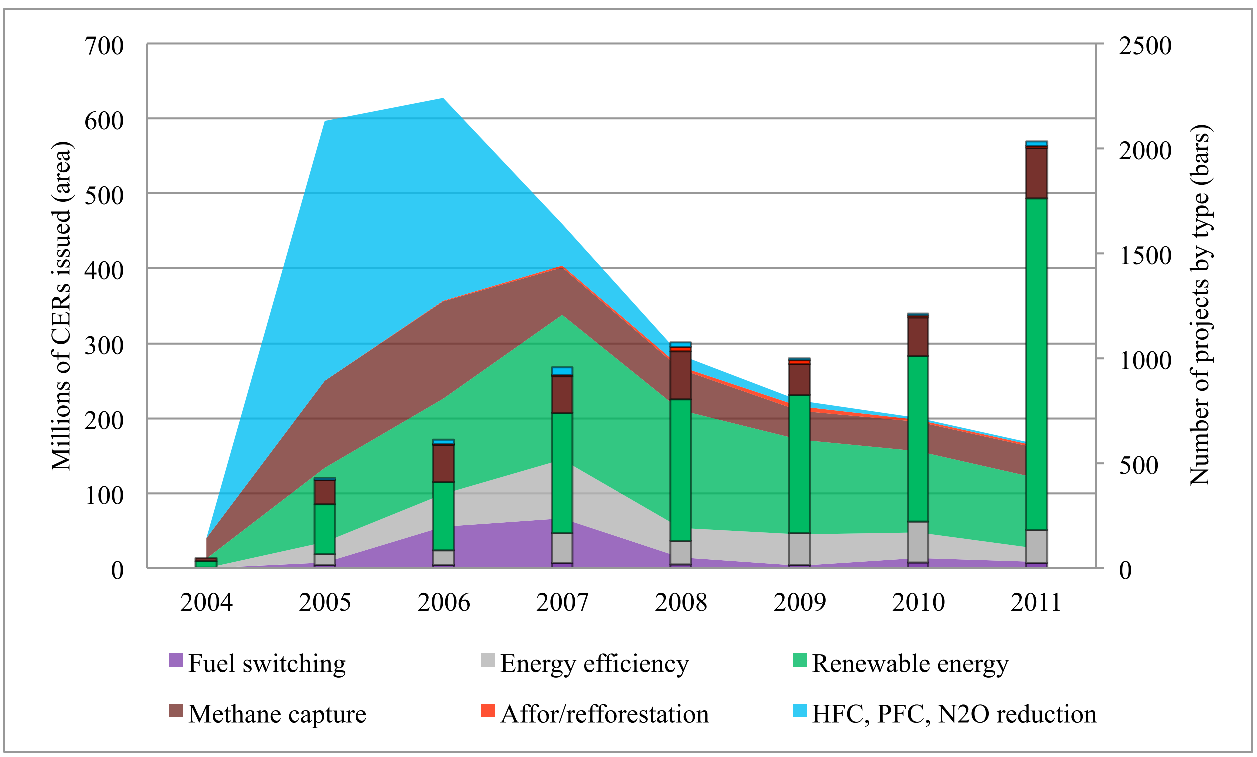
Note: Exchange-traded prices are through June 30, 2012 as reported by the following sources: Point Carbon, RGGI COATS, and Ecosystem Marketplace/Bloomberg New Energy Finance.

Figure 3. Volume of CO_2 Allowance Trades (Daily Average)



Note: Exchange-traded volumes are through June 30, 2012 as reported by the following sources: Point Carbon, RGGI COATS, and Ecosystem Marketplace/Bloomberg New Energy Finance.

Figure 4. Number of CDM Projects (Bars) and Credits Issued (Area), by Project Type



Note: Data are from Fenhann (2012). Up to 3 percent of 2011 projects may be pending validation.

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