

Reflections—Carbon Pricing in Practice*

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Introduction

Carbon pricing is a broad term that encompasses two policy approaches: emissions trading and carbon taxation. Emissions trading places a cap on the aggregate emission level and allows the market to determine the price, whereas carbon taxation sets the price and allows the market to determine the aggregate level of emissions.

Although programs to address climate change based on pricing carbon are relatively new, programs to price pollution more generally are not. Various forms of emissions trading and pollution fees (or taxes) have been around for some time (see Table 1). Progress on carbon pricing has been stalled at the international level and at the national level in the United States, but, as shown in Table 1, advances are being made elsewhere. Discussions are also under way to link some of the emissions trading systems as well as to bring new members into the fold.

Existing carbon pricing programs offer a wealth of experience from which we can draw insights about the effectiveness of particular design options and how and why context can be important. This “Reflections” reviews these operating programs and identifies some of the chief lessons from this experience. The next section provides some context through a brief look at five carbon-pricing programs. This is followed by discussions of the effects of carbon pricing programs and the lessons learned about program design. The final section offers some concluding reflections about carbon pricing.

Providing Context: A Brief Look at Five Illustrative Carbon Pricing Programs

Many options are available for designing carbon pricing programs. This section discusses five existing programs to illustrate the range of design choices.

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Table 1 Selected existing or scheduled air pollution fee or emissions trading systems

Emissions trading	Air emission fees
Traditional pollutants	Traditional air pollutants
The US Lead Phase-Out Program (lead, 1985)	Japan (sulfur oxides, 1968)
The US Sulfur Allowance Program (SO ₂ , 1990)	China (multiple pollutants, 1982)
Santiago, Chile (particulates, 1992)	France (multiple pollutants, 1985)
California Regional Clean Air Incentives Market [RECLAIM] (NO _x and SO _x , 1994)	Sweden (nitrogen oxides, 1992)
Eastern US NO _x Budget Program (NO _x 2003)	Taiwan (multiple pollutants, 1996)
Climate change	Climate change
The Kyoto Protocol's Clean Development and Joint Implementation Mechanisms (2005)	Finland (1990)
The European Union Emissions Trading Scheme (2005)	Sweden (1991)
Regional Greenhouse Gas Initiative US (2009)	Norway (1991)
New Zealand (2010)	United Kingdom (2001)
California, US (2013)	Denmark (2005)
Quebec, Canada (2013)	Alberta, Canada (2007)
Australia (2012)	Switzerland (2007)
Peoples Republic of China Pilot Programs (2013) and National Program (2015)	British Columbia, Canada (2008)
South Korea (2015)	India (2010)
Vietnam (2020)	Australia (2012)
	Japan (2012)

Swedish Carbon Tax Program

In Sweden, carbon is priced both directly through a tax on each emitted unit of carbon dioxide (CO₂) and indirectly through an energy tax on fossil fuels. The carbon tax was introduced in 1991 to complement the existing system of energy taxes, which were simultaneously reduced by 50 percent.

With the introduction of the European Union Emissions Trading System (EU ETS) in 2005, some sectors were covered by both the carbon tax and the EU ETS. To avoid double regulation, the government exempted industries covered by the EU ETS from the carbon tax. Hence all sectors are covered by a carbon price, but that coverage varies greatly across firms and sectors, with some specific activities fully exempted from carbon taxes (Jamet 2011). Neither energy nor carbon taxes are applied on electricity production, but households face a special electricity consumption tax (Johansson 2000).

British Columbia Carbon Tax Program

Since 2008, British Columbia has imposed a carbon tax on each metric tonne of carbon dioxide equivalent (CO₂-e) emissions from the combustion of fuel. This program affects an estimated 77 percent of British Columbia's total greenhouse gas (GHG) emissions.

Under this program, CO₂-e is defined as the amount of CO₂, methane, and nitrous oxide (N₂O) released into the atmosphere, with the methane and N₂O emission levels adjusted to a CO₂-e basis that accounts for their relative impact on global warming. Certain fuels, such as fuel for commercial aviation and ships, are exempted. The carbon tax is applied and collected at the

wholesale level. This approach uses the channels established earlier for collecting motor fuel taxes, which has made administration easier. The tax is ultimately passed forward to consumers via higher prices.

The carbon tax is revenue neutral (i.e., all revenue generated is returned to British Columbians through cuts in other taxes). To help protect low-income households, the Low Income Climate Action Tax Credit program provides adult residents with lump-sum tax credits that are reduced by 2 percent of net family income above specified income thresholds.

European Union Emissions Trading System

Launched in 2005, the EU ETS operates in 30 countries¹ and is the largest emissions trading system in the world. The program establishes a cap on the total amount of certain GHGs that can be emitted from installations covered by the system.² Under this cap, companies receive emission unit allowances that they can sell to, or buy from, one another as needed. At the end of each year each company must surrender enough allowances to cover all its emissions or pay penalties on any excess. Companies can bank any spare allowances to cover their future needs or to sell. The cap (and hence the number of allowances) is reduced over time so that total emissions will fall. Emissions are targeted to be 21 percent lower in 2020 than in 2005. The installations currently covered by the EU ETS account for almost half of the EU's CO₂ emissions and 40 percent of its total GHG emissions.

Although the system currently covers only CO₂ emissions, its scope will soon be expanded to include other sectors and other GHGs such as N₂O and perfluorocarbons. Pending the outcome of legal challenges, airlines are expected to be brought into the program.³ Coverage of the petrochemicals, ammonia and aluminum industries, and additional gases is expected in 2013. In 2013, the program will also begin to auction off allowances and the EU will take over the responsibility for allocation from member states.

Regional Greenhouse Gas Initiative

In 2009, ten states in the northeastern United States launched the country's first carbon-pricing program. Under the Regional Greenhouse Gas Initiative (RGGI), each participating state caps CO₂ emissions from power plants, allocates CO₂ emission allowances, and (with some exceptions) invests the proceeds in programs that further reduce emissions.

Most allowances are auctioned off, and RGGI states allocate a large proportion of the revenue from these auctions to promote energy efficiency. RGGI has a price floor, which is indexed to remove the effects of inflation, and (due to the surplus of allowances created by the large emissions reductions relative to the cap) that floor has been binding. In fact, the floor has

¹These are the twenty-seven EU Member States plus Iceland, Liechtenstein, and Norway.

²The EU ETS covers CO₂ emissions from power stations, combustion plants, oil refineries, and iron and steel works, as well as factories making cement, glass, lime, bricks, ceramics, pulp, paper, and board.

³In 2012, the EU announced that it would freeze for a year its rule that all airlines must pay for their carbon emissions for flights into and out of EU airports. Flights within the EU will still have to pay for their carbon emissions.

played a significant role because revenue from the auctions is an important source of funding for the region's popular energy efficiency programs. Without that floor the amount of funding available would have been considerably less. Energy efficiency investments incentivized by RGGI revenue have substantially reduced the energy costs of a number of large industrial facilities and appear to have made them more competitive (RGGI, Inc. 2011).

Australian Hybrid System

The Australian program (Commonwealth of Australia 2011), which started in 2012, envisions a two-stage transition from a fixed-price regime to an emissions trading market:

- In the first stage (July 1, 2012, to June 30, 2015), emitters face a fixed price for each metric tonne of carbon emitted. The price started at \$A23 (US\$23.8) per metric tonne and will rise at 2.5 percent per annum in real terms.
- On July 1, 2015, the fixed carbon price regime will transition to a fully flexible price regime, with the price determined by an emissions trading market.

While at least half of a source's compliance obligation must be met through the use of domestic permits or credits, during the flexible price period offsets from credible international carbon markets and emissions trading systems may be used to fulfill the remaining obligation.

The original design included a price ceiling and floor for the first three years of the flexible price period, but the price floor was ultimately dropped from the program due to pressure from the business community and negotiations with the EU over the conditions for linking the two systems.

Although the scope of coverage will be quite broad, it will not be universal. Four of the GHGs included in the Kyoto Protocol—CO₂, methane, N₂O, and perfluorocarbons from aluminum smelting—will be covered. Households, on-road business use of light vehicles, and the agriculture, forestry, and fishery industries will not face a carbon price on the fuel they consume but will continue to pay fuel excise taxes.

More than 50 percent of the potential revenue generated between 2011 and 2015 is targeted for reducing the cost burden on households. The plan also sets aside some revenue for assisting highly impacted firms.

What Have Been the Effects?

How have these designs worked in practice? This section examines the cost savings, emission reductions, market transformation, and technological innovation and diffusion resulting from pollution pricing in general and carbon pricing in particular.

Cost Savings

Two types of studies have conventionally been used to assess cost savings: *ex ante* analyses, based on computer simulations, and *ex post* analyses, which examine actual implementation experience.

A substantial majority, although not all, of the large number of *ex ante* analyses of programs addressing pollutants other than carbon have found that a change from more traditional regulatory measures based on source-specific limits to more cost-effective market-based measures such as emissions trading or pollution taxes could potentially achieve either similar reductions at a much lower cost or much larger reductions at a similar cost (Tietenberg 2006). The evidence also suggests that these two instruments typically produce more emissions reduction per unit expenditure than other types of policies such as renewable resource or biofuel subsidies (Productivity Commission 2011).

Although the number of existing detailed *ex post* studies is small, they typically find that the cost savings from shifting to these market-based measures are considerable, but less than would have been achieved if the final outcome had been fully cost effective. In other words, while both taxes and emissions trading are fully cost effective in principle, they fall somewhat short in practice.

Both tax and emissions trading outcomes can be distorted through political manipulation, but emissions trading is uniquely susceptible to price manipulation arising from market power that could, in principle, reduce the cost savings. However, actual experience with emissions trading has uncovered only one case of market power, which resulted directly from a design flaw. Evidence from the Regional Clean Air Incentives Market (RECLAIM), an emissions trading program in California, indicates that some generators manipulated NO_x emission permit prices in late 2000 and early 2001 (Kolstad and Wolak 2008).

The paucity of cases involving market power is perhaps not surprising since most carbon markets have a large number of participants and market power declines with large markets. However, if emissions trading expands to settings where the market is fragmented (and hence limited to relatively few participants), more cases of market power could arise.

Emission Reductions

While the evidence generally suggests that implementing these market-based programs reduces emissions (sometimes substantially), that evidence is less solid than the evidence on costs. Almost all of the emissions evidence is based on what happens to emission levels following the introduction of the program relative to what they were prior to the program. However, evaluating emissions patterns over time is problematic in at least two important ways.

- First, an historic baseline can be a highly inaccurate benchmark. Suppose, for example, that in the absence of the program the emissions level would have increased dramatically over time. This means that a program that stabilizes emissions would be judged incorrectly to have accomplished nothing, since actual emissions would not have declined relative to the baseline. In fact, however, the program would have reduced emissions substantially relative to what would have happened otherwise.
- Second, with the exception of the sulfur allowance, RECLAIM, and lead phaseout programs in the United States (where the evidence is compelling), the degree to which credit for these after-implementation reductions can be attributed solely to the market-based mechanisms (as opposed to exogenous factors or complementary policies) is limited.

With these cautionary notes in mind, under earlier (noncarbon) pollution pricing programs, emissions have generally fallen substantially following the introduction of market-based mechanisms.⁴

Impacts of carbon taxes

Sumner, Bird, and Dobos (2011) find that emission reductions resulting from carbon taxes are typically in the high single digits.⁵ However, the experience has not been uniform. Norway actually reported an emissions increase, apparently due to extensive tax exemptions and relatively inelastic demand in the sectors in which the tax was implemented (Bruvoll and Larsen 2004). Possibly in reaction to this experience, Norway is nearly doubling the CO₂ tax rate for its offshore oil and gas production in 2013.

Sweden's carbon tax appears to have caused emission reductions mainly in the residential sector (largely by encouraging district heating) and has diminished the historic trend of increasing emissions in transport. Experts believe that the carbon tax's impact on Swedish industry is probably small due to the many exemptions (Johansson 2000).

In an econometric study that attempts to control for other factors that could affect emissions outcomes, Lin and Li (2011) compare the change in per capita CO₂ emissions over time between countries that do and do not use a carbon tax. They find that (with the exception of Norway) carbon taxes have reduced emissions, but the role of carbon taxes was statistically significant only for Finland. The authors attributed this lack of statistical significance for the other countries to tax exemption policies on certain energy intensive industries.

Impacts of emission trading

The EU ETS reported "reduced annual emission per covered installation" of 8 percent from 2005 to 2010 (Hedegaard 2011). RGGI emissions from all covered sources were reduced 25.6 percent during the same period despite a rather weak cap. While the recession played some role, the main source of the reductions was fuel substitution, aided considerably by lower natural gas prices. In the RGGI region between 2005 and 2010, electricity generation from residual fuel oil fell 95 percent, generation from coal decreased 30 percent, and natural gas generation was up 35 percent (ENE 2011).

The special role of natural gas

The RGGI experience highlights an important aspect of current carbon reductions—the role of natural gas. The advent of fracking, a process combining horizontal drilling with hydraulic fracturing of shale rock, has resulted in large increases in the availability of relatively low-cost natural gas. Carbon emissions decline as electric generators substitute this now abundant gas for coal. While thus far the impact of this decline has been felt mostly in the United States, it is likely to spread, particularly to other countries with large deposits of gas-bearing shale such as Poland and China.

⁴For example, emissions from covered sources fell 67 percent in the US sulfur allowance program and lead emissions from gasoline were eliminated (see sulphur allowance program web site, http://www.epa.gov/air/markets/progress/ARP09_1.html#SO2 (accessed May 19, 2013), and Tietenberg 2006).

⁵For example, from 2008 to 2010, British Columbia's per capita GHG emissions declined by 9.9 percent (Elgie 2012).

The widespread availability of low-cost natural gas is not an unambiguous victory for the climate or for the environment in general. Methane, which is a powerful GHG, has been found to leak from these wells. These leaks offset to some degree the carbon advantages of combusting natural gas rather than coal. Fracking wells can also be a source of water contamination and local air pollution and can use large quantities of water. Finally, the International Energy Agency (2012) has shown that relying on natural gas would not be sufficient to reach the carbon reduction targets put forward in international forums; it is, after all, still a fossil fuel that releases carbon when burned.

Another concern is that low natural gas prices might impede the adoption of renewables such as wind or solar because they are substitutes in both electrical generation and heating. Government policies such as feed-in tariffs, renewable portfolio standards, and production subsidies have all played a role in the increased market penetration of solar and wind over the last few years. However, the political durability of these policies is far from certain in light of the cheap natural gas alternative.

A similar concern has been raised about the impact of natural gas prices on the market for energy efficiency. Clearly, the value of an energy efficiency investment rises when the displaced energy is expensive and falls when it is cheaper. Thus cheaper gas could well undermine some of the demand for energy efficiency. For the moment, the importance of these factors remains an open question.

Carbon leakage concerns

Of course, not all GHG sources end up being regulated, which raises the specter of leakage. Leakage can occur when pressure on the regulated source to reduce emissions results in a diversion of emissions to unregulated, or less regulated, sources. Common examples include firms moving their polluting factories to countries with lower environmental standards or consumers increasing their reliance on imported products from countries with unregulated sources. When such diversions occur, a program's *net* emissions reductions (i.e., reductions from the regulated sources minus the offsetting increases from the less regulated sources) could be smaller than its more apparent gross effects.

Generally, however, the evidence to date suggests rather small carbon leakage effects (Barker et al. 2007; IPCC 2007, 81). Even in the RGGI, where one might expect leakage to be high due to the seamless electrical grid connections between controlled and uncontrolled states, no evidence of leakage was found after the first two years of operation (RGGI Inc. 2012). The major challenge to this evidence is likely to come from increasing exports of US coal.

Should leakage become a problem, it can be controlled, to some extent, by border adjustment mechanisms, such as import tariffs or requiring importers to buy carbon allowances. However, these mechanisms have been subject to domestic court challenges.⁶ Attempts to control leakage across international boundaries using border adjustments have also faced legal challenges from the World Trade Organization (Pauwelyn 2012). As noted earlier, this has been an issue in the EU's attempt to make airlines flying into or departing from the EU subject to the EU ETS

⁶For example, in 2011, a US District Court found that the California Low Carbon Fuel Standard, which applies to transportation fuels from both within and outside the state, violates the Commerce Clause of the US Constitution by impermissibly interfering with, and discriminating against, interstate commerce.

regardless of their country of origin. Only time will tell the ultimate outcome of this conflict between trade and environmental objectives.

Market Transformation

Recent experience also provides some evidence on the behavior of this rather unique type of market. Although economic theory treats markets as if they emerge spontaneously and universally as needed, in practice participants in new or unfamiliar markets frequently require some experience with the program before they fully understand (and behave effectively in) it (Hintermann 2010; Tietenberg 2006). Both regulators and environmental managers appear to have experienced considerable “learning by doing” effects in response to programs that price pollutants. This means that markets tend to operate much more smoothly after they have been in existence for some time.

One recent trend in market transformation is the linking of previously separate systems. For example, Australia and the EU plan to link their GHG emissions trading systems starting in 2015, and California plans to link its emissions trading program to Canadian provinces through the Western Climate Initiative.

The forming and linking of regional trading systems are positive steps in a bottom-up approach to achieving the cost-effective ideal—a single, geographically comprehensive global carbon pricing system with a uniform carbon price. However, not only is achieving this ideal far from inevitable, given the slow and erratic pace of the process to date, a single global carbon market seems a long way off, even under the best of circumstances.

Technological Innovation and Diffusion

The literature contains some empirical support for the theoretical expectation that the implementation of market-based mechanisms will induce both emission-reducing innovation and the adoption of new emission-reducing technologies. While the gains in innovation from emissions trading programs have not always been as large as expected, most studies do find a statistically significant response (Bellas and Lange 2011). Furthermore, with respect to the international transfer of mitigation technologies, some evidence suggests that the Clean Development Mechanism (CDM), one component of international emissions trading, has hastened diffusion (Dechezleprêtre, Glachant, and Ménière, 2008), although others have questioned the quantitative importance of this channel (Das 2011).

Some case studies also provide evidence that environmental taxes have made a difference in promoting innovative strategies. For example, Sterner and Turnheim (2009) found that the Swedish Nitrogen Charge promoted both innovation (improvement of best practices) and diffusion (the spread of the new technologies to other firms). Johansson (2000) reports that during the 1990s, as the demand for biofuels increased in Sweden (in part due to the energy and carbon taxing system), several new more efficient wood-handling technologies were adopted.

However, the finding that market-based instruments hasten innovation and diffusion is not universal. In some circumstances, pollution pricing may encourage the exploitation of existing low-cost options (such as switching to existing lower carbon fuels or utilizing supplies of relatively cheap offsets) rather than stimulating the adoption of new technologies, thereby actually delaying their commercialization (Taylor, Rubin, and Hounshell 2005). In these

cases, the stimulus for more fundamental innovation takes place over a longer time frame, once the existing cheaper opportunities have been exhausted.

Lessons for Program Design

Countries considering carbon-pricing systems have several design options. In many cases, the choices are similar regardless of whether the preferred instrument is a carbon tax or emissions trading, but in others the choices are unique to the instrument. This section summarizes some of the key design choices and the lessons learned about preferred options from actual experience with carbon pricing systems.

Instrument Choice

In the past, public discourse frequently framed instrument choice as deciding which instrument was superior, carbon taxes or emissions trading. Now that we have several years of experience with these systems, it has also become quite common to consider how they can be combined effectively.

For example, carbon taxes and emissions trading can be used sequentially, where one instrument is used initially until a transition to the second instrument is completed. As discussed earlier, sometimes emissions trading markets take time to mature. To ensure that early markets do not produce volatile or unstable prices, one can start with a tax regime that produces known stable prices until such time as participants become familiar with abatement choices and their costs, so that an emissions market can take over. For example, as noted earlier, Australia imposed a fixed price for permits in mid-2012, which is expected to pave the way for a transition to a carbon-trading system where prices are determined by the market by mid-2015.

The two instruments can also be used effectively at the same time but applied to different sectors. It is no coincidence that emissions trading systems such as the EU ETS and the RGGI target larger sources while taxes have been typically targeted to more diffuse sources such as residential or transport emissions. The Australian emissions trading plan is another example where sources not covered by the cap are controlled by a separate, but equivalent, carbon price. Similarly, in Sweden, household and transport emissions are controlled largely via taxation while large enterprise emissions are controlled via the EU ETS.

Using Revenue from Taxes or Auctions

Carbon taxes and auctioned allowances not only provide incentives for reducing emissions, but they also raise revenue. The distribution of the revenue from auctioned allowances or carbon taxes can, in principle, enhance policy efficiency, make the distribution of the financial burden less regressive, and/or improve the political feasibility and stability of the program. However, the extent to which such benefits can be captured depends on how the revenue is used.

Targeted cost containment

Countries or regions establishing carbon-pricing programs often face powerful political resistance, especially from energy-intensive businesses, because of their possible economic impacts. These concerns have resulted in several design strategies that use potential or actual revenue to

contain possible cost increases faced by these businesses. Such strategies include the use of exemptions, differential tax rates, rebates, and gifted allowances.

Granting exemptions is a common strategy for targeted cost containment. Types of exemptions include (1) exempting all emissions from sources that emit fewer emissions than some established threshold (a strategy followed by most programs); (2) exempting emissions from sources that are covered by another policy to prevent double exposure (also common); or (3) exempting emissions from sources deemed unacceptably vulnerable to cost increases. While the first two types of exemptions may not raise serious cost-effectiveness issues, the third type does. Because facilities that receive exemptions from the policy instrument face *no* controls on GHGs, their incentive to reduce emissions is eliminated. Furthermore, when policies are designed to achieve specific quantitative targets, other facilities must pick up the slack created by exemptions, thereby raising compliance costs for the program as a whole.

Although exemptions are common in European carbon tax systems, differential tax rates are even more common (Andersen 2009). In Norway, for example, the pulp and paper industry, fishmeal industry, domestic aviation, and domestic commodity shipping all pay reduced rates. In Sweden, manufacturing, agriculture, cogeneration plants, forestry, and aquaculture pay proportionally lower rates (Sumner et al. 2011).

The Swedish Nitrogen Charge took a rebate approach to cost containment, which from the beginning was intended to provide a significant incentive rather than to raise revenue. The charge rate is high by international standards (thereby enhancing its incentive effect), but the revenue from this tax is not retained by the government. Instead it is rebated to the emitting sources (thereby reducing the impact of the tax on competitiveness).

It is the form of this rebate that makes this an interesting scheme. While the tax is collected on the basis of emissions, it is rebated on the basis of energy production. In effect, this system rewards plants that emit little NO_x per unit of energy and penalizes plants that emit more NO_x per unit of energy. Although this approach provides incentives to reduce emissions per unit of energy produced, it provides no incentives to reduce energy use. Hence it reduces fewer total emissions than an unrebated tax.

Another strategy is gifting, which can be used in either a tax system or emissions trading. In a tax system, gifting involves taxing only those emissions that are above a specified threshold. Alternatively, in a cap-and-trade system, some proportion of the allowances can be gifted (given free of charge) to favored sectors. Either approach eliminates the financial burden associated with paying for gifted emissions, but in contrast to a total exemption, gifting does not relieve the sector of its obligation to control GHGs.

Because it has become widely recognized that gifted allowances tend to exacerbate the inequities associated with carbon pricing and reduce the economic efficiency of the system, the use of gifting has decreased. For example, gifting plays a very small role in the RGGI, with approximately 86 percent of CO₂ allowances offered at auction and approximately 4 percent offered for sale at a fixed price. Even systems that grant gifted allowances tend to diminish the proportion of gifted allowances over time. Initially, both the EU ETS and California's emissions trading program gifted some or all of the allowances to parties based on some specified eligibility criteria. In the EU, free allowances were allocated based on product-specific benchmarks for each relevant product. As the California scheme begins, some 90 percent of 2008 electricity sector emissions will be covered by gifted allowances, but this allocation is slated to decline to 85 percent of 2008 output by 2020, the final year of the program (CARB 2010).

Basing the amount of gifted allocation simply on historical emissions is inferior from an efficiency standpoint because it can end up rewarding sources that have the poorest historical track record. Furthermore, if this method of gifting is known in advance, it can even discourage early reductions, lest such reductions lower the subsequent gifted allocation.

Experience with the EU ETS has enriched our understanding of the dynamics of gifted systems. Empirical evidence (mainly from the United Kingdom, the Netherlands, Germany, and the Nordic countries) indicates that in deregulated electricity markets, a significant share of the value of the gifted allowances in the marketplace was passed through to consumers in the form of higher prices (Sijm, Neuhoff, and Chen 2006). Since the allowances were gifted, the benefiting firms earned what are widely perceived as “windfall profits.”

In general, experience with gifted allocations suggests that although they are sometimes politically necessary, they have very large opportunity costs. Not only do they tend to make the carbon-pricing program less efficient, they also make the financial burden associated with compliance more regressive. Moreover, the evidence suggests that the revenue necessary to fully protect sectors that are truly vulnerable is a fraction of the total that would be derived from any revenue-raising approach (Bovenberg and Goulder 2001).

The double dividend and distributional purposes

Countries have made very different choices about how to use the tax and auction revenues. In Europe the energy and carbon taxation schemes in several EU member states have been guided by environmental tax reform (EEA 2005). This reform of national tax systems seeks to shift the tax burden from conventional sources, such as labor and capital, to alternative sources such as environmental pollution. In principle, this shift helps to create what is known as a “double dividend” because the carbon pricing reduces harmful GHGs (the first dividend) while the revenue can be used to reduce taxes on items that distort the economy and reduce income (the second dividend). The evidence suggests that this revenue allocation choice can simultaneously boost the efficiency of the program while reducing at least to some degree (depending on the choice of which distortionary taxes to reduce) the regressivity of the distributional burden of the costs (Parry et al. 2006).

Sweden and Finland have mainly recycled revenue by lowering income taxes.⁷ In contrast, Denmark and the United Kingdom have used revenues primarily to lower employers’ social security contributions (Andersen 2009). British Columbia has used the revenue primarily to lower personal, corporate, and small business income taxes.

An MIT study (Rausch and Reilly 2012) has suggested that this use of revenue could possibly break the political logjam for a national carbon pricing system in the United States by providing a win-win-win solution to the problem of simultaneously reducing both emissions and debt. “The first win—Congress could reduce personal or corporate income tax rates, extend the payroll tax cut, maintain spending on social programs, or some combination of these options. The second win—these cuts in income taxes would spur the economy, encouraging more private spending and hence more employment and investment. The third win—carbon dioxide (CO₂) pollution and oil imports would be reduced” (p. 1).

⁷While Finland does not earmark the carbon tax revenue, the increased revenue has been accompanied by independent cuts in income taxes.

Although the cost burden associated with carbon pricing without any redistribution of the revenues is apparently not regressive in developing countries (Stern 2012), it is quite regressive in developed countries (Parry et al. 2006). Since efficiency-enhancing strategies, such as lowering corporate taxes, do little to diminish that regressivity, some programs specifically target a proportion of the funds to reduce the cost burdens on households, particularly low-income households. For example, in Australia, more than 50 percent of the revenue will be used to reduce the cost burden on households.

Promoting renewable energy and energy efficiency

Some programs use the revenue to promote additional emissions reductions, a strategy that can also reduce the allowance price in an emissions trading program by reducing the demand for allowances. This strategy can lower the cost of carbon abatement and reduce the program's impact on the local economy. In Denmark, for example, although about 60 percent of the revenue is returned to industry, some 40 percent of tax revenue is used for environmental subsidies. Quebec deposits its carbon tax revenue into a "green fund" that supports measures offering the largest projected reduction in, or avoidance of, GHGs (Sumner et al. 2011). As noted earlier, the RGGI also tends to direct most of its revenue to promoting energy efficiency.

General Cost Containment: The Role for Offsets

Exemptions, preferential tax rates, rebates, and gifting all attempt to reduce the cost for certain targeted sources. Another option—offsets—reduces the compliance cost (but in this case on *all* participants) by expanding the supply of reduction possibilities.

Offsets allow emissions reductions from sources not covered by the cap or not included in the base of a carbon tax to be credited against the cap or tax base of the acquiring party. Offsets or offset tax credits perform several roles in pricing GHGs:

- First, by increasing the number of reduction opportunities, they lower the cost of compliance. The cost effects can be dramatic. For example, preliminary estimates by the US Environmental Protection Agency (USEPA 2009) suggest that if the American Clean Energy and Security Act of 2009 had become law, its liberal offset provisions would have had the effect of reducing the allowance price by approximately 50 percent.
- Second, lowering the cost in this manner could increase the likelihood of enacting a carbon-pricing program by making compliance easier.
- Third, offsets extend the reach of a program by providing economic incentives for reducing sources that are not covered by the tax or cap. Current features of the RGGI, for example, include credits for reducing methane from landfills or for the additional carbon absorption resulting from reforestation investments.
- Finally, because offset credits separate the source of financing from the source providing the reduction, it secures some reductions (in developing countries, for example) that for affordability reasons might not occur otherwise.

Offset credits may represent a transition strategy. For example, as long as some countries remain outside the cap, offset credits may represent the best opportunity to secure emission

reductions in those countries. However, once all countries fall under a cap, many current offsets will also fall under that cap and thus no longer qualify as offsets.

The challenge for establishing an effective offset program is assuring that all three of the primary requirements (namely, that the reductions be quantifiable, enforceable, and additional) are met. One obstacle is the tradeoff between transactions costs and offset validity (assuring valid offsets is not cheap). Some criticism has been aimed at the CDM, a key source of international offsets from developing countries. This criticism involves not only the types of projects being certified (an alleged overemphasis on non-CO₂ gases), but also the skewed regional distribution of activity (with China, India, South Korea, and Brazil creating more than 60 percent of generated credits). In addition, the size of the subsidy being granted has been criticized, especially since the incremental costs of reduction for some non-CO₂ gases have been considerably lower than the price received for a credit. Finally, it has been noted that the CDM creates adverse incentives for offset host countries to join an agreement that imposes a cap on their emissions. In particular, developing countries may hesitate to undertake projects on their own as long as they can get someone else to pay for them through the CDM (Hall et al. 2008).

In response to concerns over the validity of offsets, most programs now try to limit their use. One historical approach has been to restrict the use of offsets (domestic, foreign, or both) to some stipulated percentage of the total required allowances. In the RGGI, for example, CO₂ offset allowances may be used to satisfy only 3.3 percent of a source's total compliance obligation during a control period, although this may be expanded to 5 percent and 10 percent if certain CO₂ allowance price thresholds are reached. In 2011, Germany announced that it would not allow *any* offsets to be used to pursue its reduction goals, but apparently Australia plans to rely heavily on the purchase of offsets to hold costs down (Point Carbon 2011).

The disadvantage of this blunt quantitative limits approach is that it not only raises compliance costs, but it also fails to distinguish between high-quality and low-quality offsets; both are treated with the same broad brush. Newer approaches are making these kinds of quality distinctions. One option is for countries to establish eligibility criteria that identify certain offset types as acceptable and therefore eligible to be treated as fungible with allowances, while offset types where the reductions are more speculative and/or the monitoring less reliable are not allowed. For example, in 2011, Australia announced that it would not accept HFC 23 or N₂O offsets from the CDM program.

An alternative approach would be to provide a margin of safety against the uncertainty in the magnitude of the ultimate reductions from an offset project by discounting the amount of emissions reduction authorized by the offset. For example, the acquiring source might only be able to use some fraction (e.g., 50 percent) of the emissions reduction actually certified in the offset toward compliance (Sedjo and Marland 2003; Willey and Chameides 2007).

The Role of Price Volatility

A tax system fixes carbon prices and, unless some administrative intervention changes those fixed prices, price volatility is not an issue. This is not the case with emissions trading in either principle or practice.

Experience not only validates the concern that emissions trading can be plagued by volatile prices, but it also demonstrates that price volatility is not a rare event. For example, the EU ETS, RECLAIM, and US sulfur allowance program have all experienced price volatility:

- In the EU-ETS case, two early price declines were attributable to two correctable design mistakes: inadequate public knowledge of actual emissions relative to the cap and a failure to allow allowances in the first phase to be banked for use in the second phase (Ellerman 2008). A subsequent dramatic price decline in 2012 stemmed from an overallocation of permits, recession, and long-term uncertainty about climate policy.
- In the RECLAIM program, the greater Los Angeles area experienced substantial price spikes due to an unanticipated rise in the demand for allowances. This rather dramatic demand shift resulted from the unexpected unavailability of important low or nonpolluting electrical generating sources (natural gas and hydro from out of state). The effects were intensified because the program was reaching the “crossover” point (i.e., the first time that actual emissions would exceed allocations unless emission reduction controls were installed at facilities) at precisely the same time (SCAQMD 2007).
- In the US sulfur-emissions trading program, prices became volatile in the 2004–5 and 2008–9 periods. In the first period, a large rise in allowance prices was triggered by rising natural gas prices (due in part to Hurricane Katrina), while in the second period prices fell dramatically in response to two US Circuit Court rulings dealing with a different, but related, program to control sulfur: the Clean Air Interstate Rule (Burtraw and Szambelan 2009).

This experience demonstrates that the design of an emissions trading system is vulnerable to unstable prices in two rather fundamental ways:

- First, because the cap establishes a fixed supply of allowances, demand shifts (due, for example, to regulatory actions, recessions, or shifts in prices of lower carbon fuels) can trigger large changes in allowance prices (since supply cannot respond).
- Second, the demand for allowances is derived from satisfying compliance obligations. Changing circumstances (due either to external factors or simply greater than anticipated success in lowering carbon emissions) can create a surplus of allowances. This surplus may cause the price to drop precipitously since lower prices do not stimulate any increase in the quantity demanded. Both the RGGI and the EU-ETS markets experienced precisely this kind of price-dropping surplus.

One promising, but as yet untested, approach to dealing with price volatility couples a “price collar” (consisting of a safety valve price ceiling backed by an allowance reserve) with a price floor. Establishing a safety valve ceiling would allow sources to purchase additional allowances from a reserve at a predetermined price, one that is set sufficiently high to make it unlikely to have any effect unless unexpected spikes in allowance prices occur. To prevent these purchases from causing the emissions cap to be exceeded, the reserve would be established from allowances set aside for this purpose from earlier years, an expansion in the availability of domestic or international offsets, or perhaps from allowances borrowed from future allocations (Burtraw, Palmer, and Kahn 2009; Jacoby and Ellerman 2004; Murray, Newell, and Pizer 2009; Pizer 2002).

An allowance reserve has been included in the California emissions trading program and was part of the Waxman-Markey bill that passed the US House of Representatives but failed to become law. The RGGI has a price floor that is currently binding due to an excess of allowances. Australia originally intended to include a price floor, but after consulting with business, and entering talks with the EU to link the two systems, that provision was dropped.

Concluding Reflections

As a 2010 report from the National Academy of Sciences in the United States put it, “A carbon pricing strategy is a critical foundation of the policy portfolio for limiting future climate change” (NAS 2010, 6). Carbon pricing is viewed as critical not only because it fosters the transition to a low-carbon economy (and action—abatement—is ultimately cheaper than inaction—paying damages), but also because it accomplishes that goal in a more comprehensive and cost-effective way than alternatives.

The old adage “If it seems too good to be true, it probably is” certainly could apply to some descriptions of carbon pricing. Although the experience reviewed here clearly does not reveal perfection, it does suggest that when used appropriately, carbon pricing does the job and does it reasonably well.

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