Lessons Learned from Three Decades of Experience with Cap and Trade

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Introduction

Thirty years ago, many environmental advocates argued that government allocation of rights to emit pollution inappropriately legitimized environmental degradation, while others questioned the feasibility of such an approach (Mazmanian and Kraft 2009). At the time, virtually all pollution regulations took a command-and-control approach, either specifying the type of pollution control equipment to be installed or setting uniform limits on emission levels or rates.

Today, it is widely recognized that because emission reduction costs can vary greatly, the aggregate abatement costs under command-and-control approaches can be much higher than under market-based approaches, which establish a price on emissions—either directly through taxes or indirectly through a market for tradable emissions rights (called permits or allowances) established under a cap-and-trade policy. Market-based approaches tend to equate marginal abatement costs rather than emissions levels or rates across sources. This means that in theory, market-based approaches can achieve aggregate pollution control targets at minimum cost.

In this article, we examine the design and performance of seven of the most prominent emissions trading systems that have been implemented over the past 30 years in order to distill key lessons for future applications of this environmental policy instrument. We focus on systems that are important environmentally and/or economically and whose performance is well documented. We exclude emission reduction credit (i.e., offset) systems, which offer credits for emissions reductions from some counterfactual baseline, because while emissions can generally be measured directly, emissions reductions are unobservable and often ill-defined. It is worth noting, however, that offset systems have been fairly widely used, notably in the Clean Development Mechanism (CDM), an international offset system that is part of the Kyoto Protocol.

The seven emissions trading systems we examine are the U.S. Environmental Protection Agency's (EPA's) phasedown of leaded gasoline in the 1980s, the U.S. sulfur dioxide (SO_2)

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allowance trading program under the Clean Air Act Amendments of 1990, the Regional Clean Air Incentives Market (RECLAIM) in southern California; the trading of nitrogen oxides (NO_x) in the eastern United States, the Regional Greenhouse Gas Initiative (RGGI) in the northeastern United States, California's cap-and-trade system under Assembly Bill 32, and the European Union (EU) Emissions Trading System (ETS). All these programs except the first are textbook cap-and-trade systems.[1] We review the design, performance, and lessons learned from each of the seven systems, and then briefly discuss several other cap-and-trade systems. In the final section we summarize key lessons for designing and implementing new cap-and-trade systems and present our thoughts about the potential role of cap-and-trade in global climate change policy.

Experience with U.S. National Cap-and-Trade Programs

Beginning in the 1980s, the first emissions trading systems were developed and implemented at the federal level in the United States.

The Phasedown of Leaded Gasoline

In the 1970s, there was growing concern about the use of lead as an additive in gasoline. Although it was later documented that lead oxide emissions were a serious human health threat, the original concern was that these emissions were fouling catalytic converters, which were required in new U.S. cars (starting in 1975) to reduce emissions of carbon monoxide and hydrocarbons. In the early 1980s, in response to this concern, the EPA began a phasedown of lead in gasoline to 10 percent of its original level.

In 1982, the EPA launched a trading program aimed at reducing the burden on smaller refineries, which faced significantly higher compliance costs than large refineries. Unlike a textbook cap-and-trade program, in which a fixed quantity of permits is given or sold to compliance entities, there was no explicit allocation of permits. Instead, the system implicitly awarded property rights on the basis of historical levels of gasoline production (Hahn 1989). More specifically, if a refiner produced gasoline with a total lead content that was lower than the amount allowed, it earned lead "credits" that the EPA allowed the refiner to sell. Under the program's banking provision, lead credits could also be saved for later use. This created an incentive for refineries to make early reductions in lead content to help them meet the lower limits that took effect over time.

Performance

Overall, the trading program was successful in meeting its environmental targets, although it may have produced some temporary geographic shifts in use patterns (Anderson, Hofmann, and Rusin 1990; Newell and Rogers 2007), and it resulted in leaded gasoline being removed from the market faster than anticipated. In each year of the program (until the lead phasedown was completed and the program was terminated at the end of 1987), more than 60 percent of the lead added to gasoline was associated with traded lead credits (Hahn and Hester 1989). This high level of trading far surpassed levels observed in earlier environmental offset markets under

[1]See Appendix Table 1 for a brief overview of these programs.

the EPA's Emissions Trading Program in the 1970s. The level of trading and the rate at which refiners reduced their production of leaded gasoline suggest that the program was also relatively cost effective (Hahn and Hester 1989; Kerr and Maré 1997; Nichols 1997). The EPA estimated that the lead trading program resulted in savings of approximately 20 percent relative to approaches that did not include trading (U.S. Environmental Protection Agency 1985). In addition, the program provided significant incentives for cost-saving technology diffusion (Kerr and Newell 2003).

Lessons

Three major lessons emerge from the design and implementation of this program. First, as the first environmental program in which trading played a central role, the program served as proof of the concept that a tradable emission rights system could be both environmentally effective and economically cost effective.

Second, the program demonstrated that transaction costs in such a system could be small enough to permit substantial trade. In contrast, in the 1970s, the EPA's Emissions Trading Program (a set of emissions reduction credit systems) required prior government approval of individual trades, which hampered trading activity. The lack of such requirements was an important factor in the success of trading in the lead phasedown program (Hahn and Hester 1989).

Third, as in later programs, banking played a very important role. By enabling intertemporal substitution, provisions that allowed firms to bank permits contributed a significant share of the gains from trade.

The Sulfur Dioxide Allowance Trading Program

During the 1980s, there was growing concern that acid precipitation—due mainly to emissions of SO_2 from coal-fired power plants—was damaging forests and aquatic ecosystems (Glass, Glass, and Rennie 1982). However, because the costs of emissions reductions differed dramatically among existing plants, legislative proposals to use command-and-control approaches failed to attract significant support.

Title IV of the Clean Air Act Amendments of 1990 addressed this issue by launching the SO_2 allowance trading program. Phase 1 (1995–1999) required emissions reductions from the 263 most polluting coal-fired electric generating units (larger than 100 MW), almost all of which were located east of the Mississippi River. Phase 2, which began in 2000, placed an aggregate national emissions cap on approximately 3,200 electric generating units (larger than 25 MW), nearly the entire fleet of fossil-fueled plants in the continental United States (Ellerman et al. 2000). This cap represented a 50 percent reduction from 1980 levels.

The government gave power plants permits to emit (called "allowances") specific tonnages of SO_2 emissions; allocations were based primarily on actual fuel use during the 1985–1987 period.[2] If annual emissions at a regulated facility exceeded its allowance allocation, the owner could comply by buying additional allowances or reducing emissions by installing pollution

[2] In addition, the statute required the EPA to withhold about 2.8% of all allocated allowances each year, sell them at an annual auction, and return the proceeds in proportion to firms from which allowances had been withheld (Ellerman et al. 2000).

controls, shifting to a fuel mix with less sulfur, or reducing production. If emissions at a regulated facility were below its allowance allocation, the facility owner could sell the extra allowances or bank them for future use. The EPA monitored emissions on a continuous basis and verified ownership of the allowances submitted for compliance.

This cap-and-trade system created incentives for facilities to reduce their SO_2 emissions at the lowest cost. Although government auctioning of allowances would have generated revenue that could have been used—in principle—to reduce distortionary taxes, thereby reducing the program's social cost (Goulder 1995), this efficiency argument was not advanced at the time. Because the entire investor-owned electric utility industry was subject to cost-of-service regulation in 1990, it was assumed that the value of free allowances would be passed on to consumers and thus not generate windfall profits for generators. Just as important, the ability to allocate free allowances helped to build significant political support for the program (Joskow and Schmalensee 1998). Since the equilibrium allocation of pollution permits (after trading has occurred) is independent of the initial allocation (Montgomery 1972)—barring particularly problematic types of transaction costs (Hahn and Stavins 2012)—the initial allocation of allowances could be designed to maximize political support without compromising the system's environmental performance or raising its cost.

Performance

The program performed exceptionally well across all relevant dimensions. SO_2 emissions from electric power plants decreased 36 percent between 1990 and 2004 (U.S. Environmental Protection Agency 2011), even though electricity generation from coal-fired power plants increased 25 percent over the same period (U.S. Energy Information Administration 2012). The program delivered emissions reductions more quickly than expected, as utilities made substantial use of the ability to bank allowances for future use. With continuous emissions monitoring and a $2,000 per ton statutory fine for any emissions exceeding allowance holdings, compliance was nearly 100 percent (Burtraw and Szambelan 2010).

Some worried that the geographic pattern of emissions would change so as to produce "hot spots" of unacceptably high SO_2 concentrations. However, the pattern of emissions reductions was broadly consistent with model predictions, and no significant hot spots were produced (Ellerman et al. 2000; Swift 2004).

The cost of the program was significantly reduced after the substantial deregulation of railroads in 1980, which caused rail rates to fall and thus reduced the cost of burning low-sulfur Western coal in the East (Keohane 2003; Ellerman and Montero 1998; Schmalensee and Stavins 2013). That being said, cost savings were at least 15 percent and perhaps as great as 90 percent of the costs of various alternative command-and-control policies (Carlson et al. 2000; Ellerman et al. 2000; Keohane 2003). In addition, there is evidence that the program reduced costs over time by providing incentives for innovation (Ellerman et al. 2000; Popp 2003; Bellas and Lange 2011). However, for a variety of reasons, the program's costs were likely not as low as they could have been (Schmalensee and Stavins 2013).

Nevertheless, the SO_2 allowance trading program's actual costs were much lower than under command-and-control regulation—if such an approach had been politically feasible. The program's goals were achieved with less litigation (and thus less uncertainty) than is typical for traditional environmental programs, because firms that found it particularly costly to reduce emissions had the option to buy allowances instead. Moreover, firms could not complain about the EPA's exercise of administrative discretion since the law gave the EPA very little discretion. However, subsequent regulatory actions, court decisions, and regulatory responses led to the virtual elimination of the SO_2 market by 2010 (Schmalensee and Stavins 2013).

The SO_2 reductions achieved benefits that were a substantial multiple of the program's costs (Burtraw et al. 1998; Chestnut and Mills 2005). However, the program's benefits were due mainly to the positive human health impacts of decreased local SO_2 and small particulate concentrations, not the ecological benefits of reduced acid deposition that were expected when the program was enacted (Schmalensee and Stavins 2013). Nevertheless, there were also significant ecological benefits (Banzhaf et al. 2006).

Lessons

Even though the conclusion of the leaded gasoline phasedown trading program preceded the beginning of the SO_2 allowance trading program by a decade, the SO_2 system was, and still is today, often celebrated as the first important cap-and-trade program. Some of the lessons from the SO_2 program reinforce lessons from the lead phasedown program.

First, putting final rules in place well before the beginning of the first compliance period provides regulated entities with some degree of certainty, which facilitates their planning and limits price volatility in early years. In the case of the SO_2 allowance trading program, this was done 2 years prior to the implementation of phase 1.

Second, as with the lead trading program, the absence of requirements for prior approval of trades reduced both the uncertainty for utilities and the administrative costs for government, and it contributed to low transaction costs and substantial trading (Rico 1995).

Third, as with the lead trading program, banking of allowances was extremely important, accounting for more than half of the program's cost savings (Carlson et al. 2000; Ellerman et al. 2000).

Fourth, when combined with unrestricted trading and banking, a robust allowance market can be fostered through a cap that is significantly below business-as-usual (BAU) emissions.

Fifth, allocation of free allowances can be very useful in building political support.

Sixth, intrasector emissions leakage from regulated to unregulated entities can be minimized, as it was in this program, by regulating all nontrivial sources.

Finally, high levels of compliance can be ensured through rigorous monitoring of emissions and significant penalties for noncompliance.

U.S. Regional and State Programs

Over time, action on emissions trading in the United States has shifted to subnational programs, including the Regional Clean Air Incentives Market in southern California, NO_x trading in the eastern United States, the Regional Greenhouse Gas Initiative in the northeast, and California's cap-and-trade system.

The Regional Clean Air Incentives Market

The South Coast Air Quality Management District, which is responsible for controlling emissions in a four-county area of southern California, launched the Regional Clean Air Incentives

Market (RECLAIM) in 1993 to reduce NO_x emissions and in 1994 to reduce SO_2 emissions from 350 affected sources, including power plants and industrial sources in the Los Angeles area that emitted four or more tons per year of either pollutant. RECLAIM replaced command-and-control regulations that were scheduled to bring the region into compliance with national ambient air quality standards (Ellerman, Joskow, and Harrison 2003).

RECLAIM Trading Credits (RTCs) were allocated for free, with initial allocations of NO_x and SO_2 RTCs based on historical peak production levels and set at 40 to 60 percent above actual emissions until the year 2000. The NO_x and SO_2 caps declined annually by 8.3% and 6.8%, respectively, until 2003, when the market reached its overall goal of a 70% emissions reduction (Ellerman, Joskow, and Harrison 2003; Hansjürgens 2011). The compliance period was a single year and banking was not allowed. An interesting aspect of this program's design was its zonal nature: trades were not permitted from downwind to upwind sources.

Performance

RECLAIM was predicted to achieve significant cost savings via trade (Johnson and Pekelney 1996; Anderson 1997). And, by June 1996, 353 program participants had traded more than 100,000 tons of NO_x and SO_2 credits, with a value of more than \$10 million (South Coast Air Quality Management District 2016). Studies have found that emissions at RECLAIM facilities were some 20 percent lower than at facilities regulated with parallel command-and-control regulations, that hotspots did not appear, and that substantial cost savings were achieved (Burtraw and Szambelan 2010; Fowlie, Holland, and Mansur 2012).

In the program's early years, allowance prices remained in the expected range of \$500 to \$1,000 per ton of NO_x. During California's electricity crisis in 2000–2001, however, some sources of electricity were eliminated, which required generation at some RECLAIM generating facilities to increase dramatically. This caused emissions to exceed permit allocations at those facilities and, in the absence of a pool of banked allowances, resulted in a dramatic spike in allowance prices—to more than \$60,000 per ton in 2001 (Fowlie, Holland, and Mansur 2012). The program was temporarily suspended and prices returned to normal levels (less than \$2,000 per ton) by 2002, with all sources rejoining the program by 2007. As of December 2015, the 12-month moving average of NO_x prices was \$1,642 per ton (South Coast Air Quality Management District 2016).

Lessons

Three lessons emerge from the RECLAIM program. First, because the RECLAIM system included an upwind and a downwind zone, with trades allowed in only one direction, the program demonstrated that appropriate design can accommodate a nonuniformly mixed pollutant and attendant concerns about potential hot spots.

A second lesson from RECLAIM, which later became important for several carbon dioxide (CO_2) cap-and-trade systems, is that an overallocation of allowances eliminates a functioning spot allowance market. Third, provisions for the banking of allowances (along with other cost-containment elements, such as price caps) can be crucial for regulated entities to achieve compliance at a reasonable cost in years in which unanticipated circumstances cause emissions to be greater than expected.

NO_x Trading in the Eastern United States

Under EPA guidance, and enabled by the Clean Air Act Amendments of 1990, in 1999 eleven northeastern states and the District of Columbia developed and implemented the NO_x Budget Program, a regional NO_x cap-and-trade system. Given the significant adverse health effects of ground-level ozone (i.e., smog formed by the interaction of NO_x and volatile organic compounds in the presence of sunlight), the goal of the program was to reduce summertime ground-level ozone by more than 50% relative to 1990 levels (U.S. Environmental Protection Agency 2004). Some 1,000 electric generating and industrial units were required to demonstrate compliance each year during the summer ozone season (May–September).

The region covered by the program was divided into upwind and downwind zones and allowances were given to states to distribute to instate sources. Sources could buy, sell, and bank allowances within limits reflecting the seasonal nature of the ozone problem. Upwind states were given less generous allowance allocations as percentages of 1990 emissions. However, trading across zones was permitted on a one-for-one basis, and the two zones made similar reductions from baseline emissions levels (Ozone Transport Commission 2003).

In 1998, the EPA issued the NO_x State Implementation Plan (SIP) Call, which required 21 eastern states to submit plans to reduce their NO_x emissions from more than 2,500 sources. The call included a model rule, which, if adopted by a state, would enable it to meet its emission reduction obligations by participating in an interstate cap-and-trade program, known as the NO_x Budget Trading Program. All affected states adopted the model rule and the trading program went into effect in 2003, replacing the NO_x Budget Program. As in the earlier program, states were given allowances to allocate to instate sources. In 2009 the NO_x Budget Trading Program was effectively replaced by the Clean Air Interstate Rule (CAIR), and in January 2015, CAIR was replaced by the Cross-State Air Pollution Rule (CSAPR).

Performance

At the outset, the NO_x Budget Program market was characterized by uncertainty, because some trading rules were not in place when trading commenced. This resulted in high price volatility during the program's first year, although prices stabilized by the program's second year (Farrell 2000). Overall, under the NO_x Budget Program and the NO_x Budget Trading Program, NO_x emissions declined from about 1.9 million tons in 1990 to less than 500,000 tons by 2006, with 99% compliance (Butler et al. 2011; Deschenes, Greenstone, and Shapiro 2012). For the 1999–2003 period, abatement cost savings were estimated at 40 to 47 percent relative to conventional regulation, which did not include trading or banking (Farrell 2000).

Lessons

Four lessons stand out from the NO_x trading program. First, in order to avoid unnecessary price volatility, which imposes unnecessary risk on affected sources and thus raises costs, all of the components of an emissions trading program should be in place well before the program takes effect.

Second, a well-designed multistate process with federal guidance can be effective in coordinating what are legally state-level goals.

Third, the history of NO_x trading in the eastern United States provides a precedent and model for expanding the coverage of a cap-and-trade system over time to include additional jurisdictions, such as neighboring states.

Fourth, states can be given the flexibility to allocate allowances among instate sources without necessarily compromising environmental goals.

The Regional Greenhouse Gas Initiative

Nine northeastern U.S. states participate in the RGGI, the first U.S. cap-and-trade system to address CO_2 emissions. The RGGI is a downstream program that focuses only on the power sector. It began in 2009 with the goal of limiting emissions from regulated sources to 2009 levels through 2014. The emissions cap was then set to decrease by 2.5 percent each year from 2015 to 2019, when the cap would have declined to 10 percent below 2009 emissions. It was originally anticipated that meeting this goal would require a reduction of approximately 35 percent below BAU emissions (13 percent below 1990 emissions).

Due to the recession and the drastic decline in natural gas prices relative to coal prices, the emissions cap quickly ceased to be binding, and it appeared unlikely to become binding through 2020. In response, in 2012, in a preplanned review of the program, the RGGI states agreed to establish a lower cap for 2014, with 2.5% annual cuts thereafter to 2019. Reflecting these economic and policy changes, allowance prices fell from approximately \$3 per ton of CO_2 at the first auction in 2008 to the floor price of \$1.86 per ton in 2010, and rose to \$5.50 per ton in 2015.

Under the RGGI program, participating states must auction at least 25 percent of their allowances and use the proceeds to invest in energy efficiency, renewable energy, and related efforts. Auctioning was required mainly to avoid the windfall profits that would generally result from free allocation of allowances in deregulated electricity markets (Sijm, Neuhoff, and Chen 2006). In practice, states have auctioned virtually all allowances.

There is a ceiling on allowance prices via a cost containment reserve, from which additional allowances were sold when auction prices reach specified levels. There is also a price floor below which allowances are not sold at auction. Any unsold allowances are permanently retired after 3 years, thus automatically tightening the cap if there is a chronic allowance surplus. This combination of a price ceiling and a price floor serves as a price collar, thus making the RGGI program somewhat of a hybrid of a cap-and-trade system and a carbon tax.

Performance

Because the cap was not binding during the program's first compliance period (2009–2011), and has been barely binding since then, the direct impact of the RGGI program on power sector CO_2 emissions has been small, at best. However, the program's auctions have generated more than \$1 billion in revenues for the participating states. Some of this revenue has been used to finance government programs aimed at reducing energy demand and hence CO_2 emissions and the demand for allowances (Hibbard et al. 2011).

Monitoring costs have been very low because U.S. power plants were already required to report their hourly CO_2 emissions under the federal SO_2 allowance trading program. The penalty for noncompliance is that entities must submit three allowances for each allowance they are short.

Because of the geographically limited scope of the RGGI system, combined with interconnected electricity markets, emissions leakage has been a significant concern (Burtraw, Kahn, and Palmer 2006). One study found that if the program were fully binding, power imports from Pennsylvania to New York could result in emissions leakage of as much as 50% (Sue Wing and Kolodziej 2008).

Lessons

Three lessons have emerged from this program. The first, which has not been lost on policymakers, is that a cap-and-trade system that auctions its allowances can generate substantial revenue for government, whether or not the system has much effect on emissions.

Second, the leakage problem is potentially severe for any subnational program, particularly a power sector program, because of the interconnected nature of electricity markets (Burtraw, Kahn, and Palmer 2006).

Third, a changing economy can render a cap nonbinding (causing allowance prices to fall) or drive allowance prices to excessive levels. This suggests an important role for price collars. In the case of the RGGI, an effective price floor was established through the use of a reservation price in allowance auctions. The price ceiling has not been tested, however, and may be less effective because of the limited size of the cost containment reserve.

California's Cap-and-Trade System

In 2006, California enacted Assembly Bill 32 (AB-32), which required the California Air Resources Board to establish a program to cut the state's greenhouse gas (GHG) emissions to 1990 levels by the year 2020. The program includes energy efficiency standards for vehicles, buildings, and appliances; renewable portfolio standards that increase renewables' share of electricity supply from 20 to 33 percent; a low-carbon fuel standard that requires refineries to reduce the carbon content of motor vehicle fuels; and a cap-and-trade system (California Environmental Protection Agency 2014).

The AB-32 cap-and-trade system began in 2013, covering all electricity sold in California, no matter where it was generated,[3] and large-scale manufacturing. The program was expanded to include fuels in 2015, thereby covering 85% of the state's emissions. The cap declines annually until 1990 emission levels are achieved in 2020. Initially, most allowances were distributed for free, with greater use of auctions over time. Banking is allowed, and regulated entities may use approved offsets of emissions reductions from forestry, dairy digestion, and ozone-depleting substances reduction to account for up to 49 percent of their emissions reductions.

A price ceiling is established by releasing allowances from a reserve when auction prices reach specified levels. A price floor is created through an auction reservation price, with unsold allowances held until the reservation price is exceeded for six consecutive months. This combination produces an effective price collar, creating a hybrid cap-and-trade and carbon tax system. In addition, the program addresses competitiveness concerns in energy-intensive, trade-exposed (EITE) industries by granting free allowances in proportion to production levels in previous periods.

[3]California imports much of its electricity from out of state. The possibility of reshuffling the contracts involved may enable substantial leakage (Bushnell, Peterman, and Wolfram 2008).

In 2014, California's system was linked to a very similar system in Quebec (Kroft and Drance 2015), with mutual recognition of allowances for trading and compliance and joint allowance auctions.

Performance and lessons

Because California's cap-and-trade system was only launched in 2013, it is too early to assess its performance, other than to note that the auction mechanisms and other design features have functioned as anticipated. Thus the lessons from the AB-32 cap-and-trade system are related to its design rather than its performance.

First, the California system has demonstrated that using an initial free allowance allocation to build political support can transition over time to greater auctioning of allowances.

Second, the California experience is a reminder of the political pressures not to use auction revenues to reduce distortionary taxes. As of May 2015, the AB-32 auctions had generated more than $2 billion and were expected to generate nearly $4 billion by the end of 2016 (California Legislative Analyst's Office 2015). Assembly Bill 1532 (2012) requires that these funds "be used to reduce GHG emissions and, to the extent feasible, achieve co-benefits such as job creation, air quality improvements, and public health benefits."

Third, as the first CO_2 (or GHG) cap-and-trade system to be essentially economy-wide,[4] California's AB-32 system has demonstrated that this approach is as feasible as less efficient approaches that treat different sectors differently.

Fourth, the AB-32 system greatly limits price volatility by employing an effective price collar. As noted earlier, although emissions levels are less certain under such hybrid systems, lower price volatility reduces compliance costs.

Fifth, California has employed an effective mechanism to address concerns about competitive impacts on EITE sectors. Granting free allowances to firms in specific sectors in proportion to their production levels in a previous time period subsidizes production and thus directly affects competitiveness. Of course, this subsidy of EITE sectors introduces its own inefficiencies. On the other hand, simply granting extra allowances to firms in EITE sectors (as in the EU's ETS) has no effect on competitiveness because marginal production costs are not affected.[5]

Sixth, California's strong interest in linking its cap-and-trade system with those in other jurisdictions (including its recent linkage with Quebec) illustrates the desirability of using such linkages to reduce abatement costs, price volatility, and market power (Ranson and Stavins 2013).

Finally, although policies that address energy-related market failures can reduce costs, California's AB-32 system illustrates that some "complementary policies" are more likely to increase costs with no effect on emissions. For example, the state's low-carbon fuel standard (LCFS) requires that California refineries produce fuel with, on average, no more than a set amount of lifecycle carbon content. But refineries and transportation fuels are already covered by the cap-and-trade system, so the LCFS cannot reduce emissions in the short run unless it makes the allowance price floor binding. Because the LCFS is a binding constraint on refiners,

[4]Since 2010, New Zealand has had an economy-wide (except for agriculture) CO_2 emissions trading system linked to international allowance markets under the Kyoto Protocol, but domestic emissions are not capped.
[5]For a review of the literature on the competitiveness benefits and the efficiency costs of output-based updating of allowance allocations, see Fowlie (2012).

refiners achieve additional CO_2 emission reductions beyond what would be achieved through the cap-and-trade system alone. However, unless the price floor becomes binding, this "complementary" policy—the LCFS—will produce 100 percent leakage to other sectors when allowances are sold. In any case, marginal abatement costs are not equated across sectors and sources,[6] so aggregate abatement costs will increase. In addition, allowance prices will be depressed, raising concerns about the ability of the cap-and-trade system to encourage technological change—except in the refinery sector. In short, the LCFS is a "complementary" policy that mainly increases abatement costs and lowers allowance prices (Goulder and Stavins 2011). Many other so-called complementary policies have similar perverse effects.[7]

THE EU Emissions Trading System

The EU ETS, a cap-and-trade system focused on CO_2, is the world's largest and first multi-country emissions trading system (European Commission 2012). The EU ETS was adopted in 2003 and covers about half of EU CO_2 emissions in 31 countries[8] (Ellerman and Buchner 2007). More than 11,000 entities are regulated, including electricity generators and large industrial sources. Competitiveness concerns were largely addressed by the allocation of free allowances to a long list of selected sectors. The EU ETS excludes most sources in the transportation, commercial, and residential sectors, although some aviation sector emissions were brought under the cap in 2012.

The EU ETS was designed to be implemented in phases: a pilot phase 1 from 2005 to 2007, a Kyoto phase 2 from 2008 to 2012, and a series of subsequent phases that are now being extended through 2030. Penalties for violations increased from €40 per ton of CO_2 in the first phase to €100 in the second phase. The first phase allowed trading only in CO_2, but the second phase broadened the program to include some other GHGs.

The allocation process was initially decentralized (Kruger, Oates, and Pizer 2007), with each member state responsible for proposing its own national cap, subject to approval by the European Commission. This created incentives for member states to set high caps (Convery and Redmond 2007).

Performance

The EU ETS has performed as might have been anticipated. In January 2005, the phase 1 allowance price per ton of CO_2 was approximately €8; by early 2006, it exceeded €30, reflecting anticipated increases in demand. However, once it became clear that the generous allocation of allowances in 2005 had exceeded actual emissions, the allowance price fell by about half during one week in April 2006, fluctuated and soon returned to about €8, and then collapsed to zero in 2007 (Convery and Redmond 2007). This volatility was attributed to the absence of good emissions data at the beginning of the program, a surplus of allowances, and energy price

[6]As of January 2016, LCFS credits were selling for an average of \$105 per ton of CO_2 (California Environmental Protection Agency 2016), while the cap-and-trade allowances were selling for about \$13 per ton of CO_2 (Climate Policy Initiative 2016).
[7]In fact, the requirement that auction revenues from the cap-and-trade system be used to further the goals of the statute (AB-32) virtually guarantees this perverse interaction between "complementary policies" and the cap-and-trade system.
[8]All 28 EU countries plus Iceland, Lichtenstein, and Norway.

volatility; the collapse was attributed to the inability to bank allowances from phase 1 to phase 2 (Market Advisory Committee 2007).

The first and second phases of the EU ETS required member states to distribute almost all of the emissions allowances for free. However, since 2013, member states have been required to auction increasing shares of their allowances. The initial free distribution of allowances led to complaints about "windfall profits" for electricity generators when electricity prices increased significantly in 2005. But higher fuel prices also played a role in the electricity price increases, and some generators' profits reflected their ownership of low-cost nuclear or coal generation in areas where the market electricity price was set by higher-cost natural gas plants (Ellerman and Buchner 2007).

The system's cap was tightened for phase 2 (2008–2012) and its scope was expanded to cover new sources in countries that had participated in phase 1 as well as countries that joined the EU in 2007 and 2013. In addition, three nonmember states— Norway, Iceland, and Liechtenstein—joined the EU ETS in 2008. Allowance prices in phase 2 increased to more than €20 in 2008 and then fell when the recession reduced energy demand, thus reducing demand for allowances. Demand also declined because of the heavy use of offsets produced under the Kyoto Protocol's CDM. By the fall of 2011, prices had fallen to €10 and have remained in the €5 to €10 range since then.

The EU ETS has been extended through its phase 3 (2013–2020) with a more stringent, centrally determined cap (20% below 1990 emissions), auctioning of a larger share of allowances, tighter limits on the use of offsets, and unlimited banking of allowances between phases 2 and 3. Free allocation of allowances continues in phase 3 for EITE sectors (Sartor, Lecourt, and Palliere 2015).

There continues to be concern in the EU regarding low allowance prices (Löfgren et al. 2015). These prices reflect the weak European economic recovery and the lack of a price floor. In addition, other binding EU policies, particularly renewable generation and energy efficiency standards, reduce emissions under the cap. As noted earlier, in the absence of a binding price floor, such "complementary" policies raise costs and reduce allowance prices without affecting total emissions.

Lessons

Five main lessons have emerged from our experience with the EU ETS thus far. First, the availability of good data is important for sound allowance allocation and cap-setting decisions. Had such data been available in phase 1 of the EU ETS, it might have been possible to avoid the overallocation that occurred.

Second, to avoid an artificial price collapse at the end of a compliance period, it is necessary to allow for banking from one period to the next. Because the EU ETS did not allow banking in phase 1, it was hardly surprising that phase 1 allowance prices fell to zero at the end of phase 1.

Third, like the AB-32 California system, the EU ETS illustrates the perverse outcomes that result when "complementary" policies are applied to reduce emissions that are also covered under the cap, particularly in the absence of a price floor. Unless such complementary policies apply to sources outside the cap or address other market failures, they relocate emissions, drive up aggregate abatement costs, and depress allowance prices.

Fourth, although granting free allowances can help address distributional concerns as well as serving other political purposes, it is ultimately insufficient for dealing with international competitiveness concerns, because unless allocations are linked to production, they do not affect marginal production costs.

Finally, the history of the EU ETS shows that it is possible to move over time from a regime of generally free allowances to one in which most allowances are auctioned.

Other Cap-and-Trade Systems

Other cap-and-trade systems have been implemented, planned, or at least contemplated in many nations. Under the 1987 Montreal Protocol, several countries implemented systems of tradable rights for ozone depleting substances (ODS) during the ODS phasedown from 1991 to 2000 (Klaassen 1999; U.S. Environmental Protection Agency 2014). In addition, an international CO_2 cap-and-trade system has nominally operated since 2008: countries with emissions reduction commitments under the Kyoto Protocol (the "Annex I countries") that have ratified the protocol can, in effect, sell emission reductions that go beyond their compliance obligations to other Annex I parties that have outstanding compliance obligations. However, because the trading agents are nations rather than firms, not surprisingly there has been little activity (Hahn and Stavins 1999). There has been more international private sector activity in emissions offsets under the Kyoto Protocol's CDM.

Currently, there are CO_2 cap-and-trade systems at various stages of development in a number of countries around the world, including Japan (Sopher and Mansell 2014a), South Korea (Park and Hong 2014), Kazakhstan (Kossoy et al. 2014), and Switzerland (Sopher and Mansell 2014b). Most importantly, China began municipal and provincial pilot trading systems in 2013 (Kossoy et al. 2014), and on September 25, 2015, President Xi Jinping announced that in 2017 China will launch a national CO_2 cap-and-trade system covering key industries (Cunningham 2015).

Cap-and-trade systems have also been proposed in other countries at levels of governance that range from submunicipal to national (Kossoy et al. 2014; Organization for Economic Cooperation and Development and World Bank Group 2015). Notably, the government of Ontario (Canada) recently announced a CO_2 cap-and-trade system to be linked to Quebec's system, and thus to California's system (Government of Ontario 2015). Finally, in August 2015, the United States finalized the Clean Power Plan (CPP), which is aimed at CO_2 emissions from electricity generators and both enables and encourages state-level and multistate emissions trading (U.S. Environmental Protection Agency 2015). However, on February 9, 2016, the U.S. Supreme Court halted implementation of the CPP, pending the resolution of legal challenges to it, and thus its ultimate fate is unclear.

Summary and Conclusions

This article has examined 30 years of experience with emissions trading systems. Overall, we have found that cap-and-trade systems, if well designed and appropriately implemented, can achieve their core objective of meeting targeted emissions reductions cost-effectively. But the devil is in the details, and design as well as the economic environment in which systems are implemented are very important. Moreover, as with any policy instrument, there is no guarantee of success.

Based on the lessons we have identified in our discussion, several design and implementation features of cap-and-trade programs appear critical to their performance.

Key Features for System Design and Implementation

First, it is important not to require prior approval of trades. In contrast to early U.S. experience with emissions offset systems, transaction costs can be low enough to permit considerable efficiency-enhancing trade if prior approval of trades is not required. Second, it is clear from both theory and experience that a robust market requires a cap that is significantly below BAU emissions. Third, to avoid unnecessary price volatility, it is important for final rules (including those for allowance allocation) to be established and accurate data supplied well before commencement of a system's first compliance period. Fourth, high levels of compliance in a downstream system can be achieved by ensuring there is accurate emissions monitoring combined with significant penalties for noncompliance. Fifth, provisions for allowance banking have proven to be very important for achieving maximum gains from trade, and the absence of banking provisions can lead to price spikes and collapses. Sixth, price collars are important. A changing economy can reduce emissions below a cap, rendering it nonbinding, or a growing economy can increase emissions and drive allowance prices to excessive levels. Price collars reduce price volatility by combining an auction price floor with an allowance reserve. The resulting hybrid systems will generally have lower costs (as more stable prices facilitate investment planning) at the expense of less certain emissions reductions. Finally, economy-wide systems are feasible, although downstream, sectoral programs have been more commonly employed.

Political Considerations that Affect Cap-and-Trade Design

Thirty years of experience with cap-and-trade also indicates the importance of political considerations for the design of cap-and-trade programs. First, because of the potentially large distributional impacts involved, the allocation of allowances is inevitably a major political issue. Free allowance allocation has proven to help build political support. And, under many circumstances, the equilibrium allowance distribution, and hence the aggregate abatement costs of a cap-and-trade system, are independent of the initial allowance allocation (Montgomery 1972; Hahn and Stavins 2012). This means that the allowance allocation decision can be used to build political support and address equity issues without concern about impacts on overall cost-effectiveness.

Free allowance allocation does forego the opportunity to cut overall social costs by auctioning allowances and using the proceeds to cut distortionary taxes. On the other hand, experience has shown that political pressures exist to use auction revenue not to cut such taxes, but to fund new or existing environmental programs or relieve deficits. Indeed, cap-and-trade allowance auctions can and have generated very significant revenue for governments.

Second, the possibility of emissions leakage and adverse competitiveness impacts has been a prominent political concern in the design of cap-and-trade systems. Of course, virtually any meaningful environmental policy will increase production costs and thus could raise these concerns, but this issue has been more prominent in the case of cap-and-trade instruments. In practice, leakage from cap-and-trade systems can range from nonexistent to potentially quite serious. It is most likely to be significant for programs of limited geographic scope,[9] particularly in the power sector because of interconnected electricity markets. Attempts to reduce leakage and competitiveness threats through free allocation of allowances does not per se address the problem, but an output-based updating allocation can do so.

Third, although carbon pricing (through cap-and-trade or taxes) may be necessary to address climate change, it is surely not sufficient. In some cases, abatement costs can be reduced through the use of complementary policies that address other market failures, but the types of "complementary policies" that have emerged from political processes have instead addressed emissions under the cap, thereby relocating rather than reducing emissions, driving up abatement costs, and suppressing allowance prices.

Identifying New Applications

Cap-and-trade systems are now being seriously considered for a wide range of environmental problems. Past experience can offer some guidance as to when this approach is most likely to be successful (Stavins 2007).

First, the greater the differences in the cost of abating pollution across sources, the greater the likely cost savings from a market-based system—whether cap-and-trade or tax—relative to conventional regulation (Newell and Stavins 2003). For example, it was clear early on that SO_2 abatement cost heterogeneity was great, because of differences in the ages of plants and their proximity to sources of low-sulfur coal (Carlson et al. 2000).

Second, the greater the degree of mixing of pollutants in the receiving airshed (or watershed), the more attractive a market-based system, because when there is a high degree of mixing, local hot spots are not a concern and the focus can thus be on cost-effective achievement of aggregate emissions reductions. Most cap-and-trade systems have been based on either the reality or the assumption of uniform mixing of pollutants. However, even without uniform mixing, well-designed cap-and-trade systems can be effective (Montgomery 1972), as illustrated by the two-zone trading system under RECLAIM, at the cost of greater complexity.

Finally, since Weitzman's (1974) seminal analysis of the effects of cost uncertainty on the relative efficiency of price versus quantity instruments, it has been well known that in the presence of cost uncertainty, the relative efficiency of these two types of instruments depends on the pattern of costs and benefits. Subsequent literature has identified additional relevant considerations (Stavins 1996; Newell and Pizer 2003). Perhaps more importantly, theory (Roberts and Spence 1976) and experience have shown that there are efficiency advantages of hybrid systems that combine price and quantity instruments in the presence of uncertainty.

Implications for Climate Change Policy

Two lessons from 30 years of experience with cap-and-trade systems stand out. First, cap-and-trade has proven itself to be environmentally effective and economically cost effective relative to

[9]For this and other reasons, linkage between cap-and-trade systems and other types of policies in other jurisdictions is likely to become increasingly important in the future because such linkage can reduce abatement costs, leakage, price volatility, and market power.

traditional command-and-control approaches. Moreover, less flexible systems would not have led to the technological change that appears to have been induced by market-based instruments (Keohane 2003; Schmalensee and Stavins 2013) or the induced process innovations that have resulted (Doucet and Strauss 1994). Second, and equally important, the performance of cap-and-trade systems depends on how well they are designed. In particular, we have emphasized the importance of reducing unnecessary price volatility and argued that hybrid designs offer an attractive option if some variability of emissions can be tolerated, since substantial price volatility generally raises costs.

These lessons suggest that cap-and-trade merits serious consideration when regions, nations, or subnational jurisdictions are developing policies to reduce GHG emissions. And, indeed, this has happened. However, because any meaningful climate policy will have significant impacts on economic activity in many sectors and regions, proposals for such policies have often triggered significant opposition.

In the United States, the failure of cap-and-trade climate policy in the Senate in 2010 was essentially collateral damage from a much larger political war that has decimated the ranks of both moderate Republicans and moderate Democrats (Schmalensee and Stavins 2013). Nevertheless, political support for using cap-and-trade systems to reduce GHG emissions has emerged in many other nations. In fact, in the negotiations leading up to the Paris conference in late 2015, many parties endorsed key roles for carbon markets, and broad agreement emerged concerning the value of linking those markets (codified in Article 6 of the Paris agreement).

It is certainly possible that three decades of high receptivity to cap-and-trade in the United States, Europe, and other parts of the world will turn out to have been only a relatively brief departure from a long-term trend toward reliance on command-and-control environmental regulation. However, in light of the generally positive experience with cap-and-trade reported here, we remain optimistic that the recent tarnishing of cap-and-trade in U.S. political debates will itself turn out to be a temporary departure from a long-term trend of increasing reliance on market-based environmental policy instruments.

Appendix Table 1 Summary of Major Cap-and-Trade Systems

System	Geographic scope	Coverage and sectors	Time period	Allowance allocation method	Cost containment mechanisms	Environmental and economic performance
Leaded gasoline phasedown	USA	Gasoline from refineries	1982–1987	Free	Banking	Phasedown completed successfully, faster than anticipated, with cost savings of $250 million per year
Sulfur dioxide allowance trading	USA	SO_2 from electric power	1995–2010	Free	Banking	Cut SO_2 emissions by half, with cost savings of $1 billion per year, but market closed due to judicial actions
Regional Clean Air Incentives Market (RECLAIM)	South Coast Air Quality Management District, CA	NO_x and SO_2 from electric power and industrial sources	1993–present	Free	—	Emissions lower than with parallel regulations; unquantified cost savings; electricity crisis caused allowance price spike and temporary suspension of the market
NO_x trading in the eastern United States	12–21 U.S. states	NO_x from electric power and industrial sources	1999–2008	Free	—	Significant price volatility in the first year; NO_x emissions declined from 1.9 (1990) to 0.5 million tons (2006); cost savings 40 to 47%
Regional Greenhouse Gas Initiative	Nine northeastern U.S. states	CO_2 from electric power	2009–present	Nearly 100% auction	Banking, cost containment reserve, auction reservation price	Cap nonbinding, then barely binding due to low natural gas prices; has generated more than $1 billion for participating states
AB-32 cap-and-trade	California	CO_2 from electric power, industrial, and fuels	2013–2020	Transitions from free to auction	Banking, allowance price containment reserve, auction reservation price	Covers 85% of emissions; reduces competitiveness effects with output-based updating (OBU) allocation; linked with Quebec cap-and-trade system
European Union Emissions trading system	27 EU Member states plus Iceland, Lichtenstein, & Norway	CO_2 from electric power, large industrial, and aviation	2005–present	Transitions from free to increased use of auctions	Banking after 2008, previous use of offsets from CDM	Overallocation by member states in pilot phase; suppressed allowance prices due to "complementary policies," CDM glut, slow economic recovery

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