The Relative Merits of Carbon Pricing Instruments: Taxes versus Trading

Robert N. Stavins*

Introduction

In 2007, the Brookings Institution's Hamilton Project asked me to write a paper proposing a trading system to reduce US emissions of carbon dioxide (CO_2). I replied that I preferred to write about what I considered to be two symmetric carbon pricing instruments: a carbon tax and a carbon trading program. The Hamilton Project leaders said that they would find someone else to write about carbon taxes (Metcalf 2007) and that I should instead focus on making the strongest possible case for what is today called "cap-and-trade." I did my best and, in the process, came to be identified as an advocate for emissions trading to address global climate change.[1] In this article, I approach the issue as I wished to more than a decade ago, treating both pricing instruments in a balanced manner and examining their advantages and disadvantages without favoring one over the other.

Nearly all industrialized countries and most large emerging economies have launched or are launching policies aimed at reducing greenhouse gas (GHG) emissions. Of the 169 parties to the Paris climate agreement that have submitted pledges to reduce their emissions, more than half indicate that they will use carbon pricing instruments. To date, 61 carbon pricing policies, including 30 carbon taxes and 31 emissions trading systems, have been implemented or are scheduled for implementation worldwide. Most of the trading systems are cap-and-trade, but an important one—China's system, implemented in 2021—is a tradable performance standard. Together, these carbon pricing initiatives will cover about 22 percent of global GHG emissions (World Bank Group 2020).

*A. J. Meyer Professor of Energy and Economic Development, John F. Kennedy School of Government, Harvard University; University Fellow, Resources for the Future; and Research Associate, NBER (email: robert_stavins@harvard.edu)

This article draws in part from Stavins (2019). Valuable comments on that working paper were provided by Lawrence Goulder, Larry Karp, Gilbert Metcalf, William Pizer, and Richard Schmalensee. I am grateful to Marika Tatsutani, two anonymous referees of this journal, and Suzanne Leonard, who provided exceptionally helpful comments and suggestions for this article. I am responsible for all remaining errors.

[1]The design recommendations in my Hamilton Project paper (Stavins 2007) served as the starting point for early efforts by the Obama administration to design a CO_2 cap-and-trade proposal. That proposal became part of the failed Waxman-Markey legislation, the American Clean Energy and Security Act of 2009.

Electronically published January 10, 2022

Review of Environmental Economics and Policy, volume 16, number 1, winter 2022.
 https://doi.org/10.1086/717773

There is widespread agreement among most economists that economy-wide carbon pricing will be a necessary (although not sufficient) component of any effective policy that can achieve meaningful and cost-effective CO_2 reductions in large and complex economies (Metcalf 2009; Kaplow 2010; Borenstein et al. 2019). Given the diversity of CO_2 sources in such economies, conventional technology and performance standards would be infeasible and, in any event, excessively costly (Newell and Stavins 2003). The key advantages of pricing instruments are their flexibility and the incentives they provide for achieving overall cost-effectiveness (Knittel 2019). Moreover, pricing approaches can reduce long-term abatement costs by inducing climate-friendly technological change (Newell, Jaffe, and Stavins 1999).

There is less agreement among economists regarding which is the better pricing instrument. Some support carbon taxes (Mankiw 2006; Nordhaus 2007), while others favor cap-and-trade (Ellerman, Joskow, and Harrison 2003; Keohane 2009). A carbon tax places a price on carbon directly—ideally upstream, where fossil fuels enter the economy. By contrast, an upstream cap-and-trade system assigns scarce permits (allowances) for quantities of carbon where they enter the economy, with the market for the allowances generating a carbon price.

How do these pricing approaches compare in terms of their anticipated and actual performance? I review and synthesize theory and experience concerning carbon taxes and cap-and-trade and find that they are equivalent—or nearly equivalent—along some important dimensions but significantly different along others. However, many of these differences fade with implementation. Indeed, what appears at first glance to be a dichotomous choice between two distinct instruments can turn out to be a choice of specific design elements along a policy continuum.

In the next two sections, I explore the theoretical similarities and differences between carbon taxes and cap-and-trade. Next, I examine the theoretical possibility and empirical reality of hybrid policy instruments that combine features of the two approaches. I then consider some of the key lessons that can be learned from experience with implementation of both types of policies. I offer a summary and some conclusions in the final section.

Similarities between Carbon Taxes and Cap-and-Trade

In theory, carbon taxes and cap-and-trade can be designed to be perfectly equivalent in terms of three attributes (incentives for achieving emission reductions, aggregate abatement costs, and effects on competitiveness), nearly equivalent in terms of potential for raising revenue, and similar in terms of costs to regulated firms and distributional impacts.

Incentives for Achieving Emission Reductions

In the absence of uncertainty, tax and trading instruments can achieve the same emission reductions, whether the point of regulation is upstream (carbon content of fossil fuels), midstream (electricity generation), or downstream (electricity use). While the point of regulation does not affect either instrument's aggregate cost, it can affect decisions about the scope of coverage (ranging from a single sector to economy-wide), which in turn has implications for environmental effectiveness as well as cost-effectiveness. Indeed, by focusing on the carbon content of fossil fuels upstream (at the mine mouth, the wellhead, and the point of import), either a tax or a cap-and-trade program could cover up to 98 percent of US CO_2 emissions. This would also limit the number of compliance entities (on the order of a few thousand), versus regulating the hundreds of millions of individual smokestacks, tailpipes, and other downstream sources that emit CO_2 (Metcalf 2007; Stavins 2007).

Aggregate Abatement Costs

If firms operate in a competitive market, either pricing instrument provides strong incentives for them to minimize their costs, which include the costs to abate emissions plus either their tax liability under a carbon tax or their costs to purchase any needed allowances (net of revenues from allowance sales) under a cap-and-trade system. Each firm faces incentives to abate emissions up to the point where the firm's marginal abatement cost is equal to the tax rate or the market-determined allowance price. Because all firms abate emissions at the same marginal cost, both instruments lead to the same (minimized) aggregate abatement cost. In principle, additional cost savings could be realized with either approach by allowing regulated sources to use offsets for emission reductions achieved outside the program to reduce tax liability or allowance requirements (Goulder and Schein 2013).[2]

Consideration of the relative effects of the two approaches on technology innovation adds a temporal dimension to the comparison. Several theoretical explorations of this issue have found that taxes and cap-and-trade (with auctioned allowances)[3] are equivalent in their ability to incentivize carbon saving innovation (Milliman and Prince 1989; Jung, Krutilla, and Boyd 1996), although under particular market conditions, carbon taxes could be superior because of their impacts on related energy prices (Scotchmer 2011).

Effects on International Competitiveness

By increasing the cost of producing carbon-intensive goods and services within a jurisdiction, a climate policy may shift comparative advantage to the production of those goods and services in jurisdictions where comparable climate policies do not exist. In theory, this can cause "leakage" of economic activity and related emissions. Such leakage may be relatively modest, particularly in internationally nontraded sectors, such as transportation and electricity production.[4] However, some energy-intensive manufacturing industries could face incentives to relocate (Aldy and Pizer 2015). Additional emissions leakage may occur through international energy markets, specifically when reduced fossil fuel consumption in countries with climate policies reduces demand and drives down fuel prices, and countries without such policies respond by increasing consumption (Aldy and Stavins 2012).

[2]Offsets refer to emission reduction credits generated when an entity reduces emissions below what they otherwise would have been. However, this raises concerns about environmental efficacy because it requires comparison with a fundamentally unobservable counterfactual.

[3]Throughout the article, I focus on cap-and-trade systems in which allowances are auctioned, both because this design maximizes the symmetry between price and quantity instruments and because most trading systems appear to be evolving in that direction.

[4]However, if the relevant jurisdiction is subnational, interconnected electricity markets can foster leakage with either instrument.

Such leakage would be equally likely to occur with either type of pricing instrument (and indeed with nearly any meaningful CO_2-limiting policy). Moreover, such leakage impacts can be mitigated, in theory, through program design. For example, border adjustments—a tax on imports of products from countries without equivalent climate policies—can be implemented together with a carbon tax (Flannery et al. 2018; Cosbey et al. 2019). In cap-and-trade systems, the imposition of an allowance requirement on imports, together with an allowance allocation that is periodically updated on the basis of product output levels, can help address competitiveness concerns (Stavins 2007; Goulder and Schein 2013). However, such allocations can have perverse effects because they tend to subsidize carbon-intensive production (Bushnell and Chen 2012).

Possibilities for Raising Revenue

An important attribute of a carbon tax is that it raises revenue for government. This revenue can be used to achieve various objectives, including reducing other distortionary taxes, which lowers the net social cost of the overall policy (Bovenberg and Goulder 1996). Of course, auctioning allowances in a cap-and-trade system can, in principle, accomplish the same objective (Goulder and Schein 2013). Although both pricing mechanisms can be nearly equivalent in their ability to raise revenue, in practice, there has been more variation in allowance prices in cap-and-trade auctions than in emission quantities in carbon tax systems, which means that tax systems may well offer greater certainty regarding government revenue (Carl and Fedor 2016).

Social Costs

The social cost of a carbon tax or a cap-and-trade system with allowance auctioning depends on how program revenues are used. Relative to a lump-sum redistribution (e.g., through direct rebates to taxpayers), recycling revenue through rate cuts in payroll taxes, individual income taxes, or corporate income taxes has been estimated to reduce the net cost of a carbon tax (after accounting for the benefits of eliminating the excess burden of distortionary taxes) by 15, 26, or 67 percent, respectively (Goulder and Hafstead 2018).

The social costs for a cap-and-trade system with complete auctioning of allowances are similar. If revenue is redistributed via a lump-sum rebate, the social cost of the policy is identical to the social cost of a carbon tax paired with a rebate. Goulder and Hafstead (2018) find that recycling auction revenue through cuts in tax rates produces "slight differences" in costs relative to making the same tax cuts using revenue from a carbon tax. But if the comparison is between a cap-and-trade system with free allowance allocation (which generates no revenue for government) and an ordinary carbon tax, then the difference in social costs is significant (Parry, Williams, and Goulder 1999).

Distributional Impacts

From the perspective of regulated firms, a carbon tax looks much like a cap-and-trade system if allowances are auctioned. Similarly, a carbon tax that includes tradable exemptions for a specified quantity of emissions (i.e., the tax is levied only on emissions above a certain threshold) can resemble a cap-and-trade system with free allowance allocation (Goulder and Schein 2013).

The distributional impacts can also be identical for households, but they depend on two design elements: the extent of "free" emissions and the use of revenue. This finding is based on an assessment of how carbon pricing affects household expenditures ("use-side" impacts) and household income (source-side impacts; Goulder and Hafstead 2018; Goulder et al. 2018; Metcalf 2019a, 2019b). Use-side impacts occur through changes in the relative prices of goods and services purchased by households, while source-side impacts occur through changes in nominal wages, capital, and transfers, which in turn affect household income.

On the use side, the effects of carbon pricing are generally regressive (ignoring the return of any revenue) because of price increases, but the degree of regressivity can be altered depending on how the revenue is used (Goulder and Hafstead 2018). On the source side, changes in wage and capital income (as well as government transfers) are generally progressive, reflecting the fact that carbon-intensive industries also tend to be relatively capital-intensive. This means that the burden of a carbon price tends to fall more on capital than on labor, and since capital income represents a larger share of income for wealthier households, the impacts are generally progressive.

If revenue from a carbon tax or allowance auction is used to lower corporate tax rates, the overall effect is still progressive, although to a somewhat lesser extent. Because source-side impacts are larger than use-side impacts for most cases in which carbon pricing is paired with revenue recycling, the overall impact of either policy—a tax or cap-and-trade with 100 percent allowance auctioning—is likely to be progressive.

Differences between Carbon Taxes and Cap-and-Trade

In terms of differences, carbon taxes and cap-and-trade are, in theory, somewhat different in terms of transaction costs, different in terms of performance in the presence of uncertainty and linkage with other jurisdictions, and significantly different in terms of carbon price volatility, interactions with complementary policies, potential for corruption and market manipulation, and complexity and administrative requirements.

Transaction Costs

Because cap-and-trade systems involve allowance trading (unless, of course, the initial allocation matches the cost-effective equilibrium), there may be transaction costs that increase aggregate compliance costs.[5] Importantly, if transaction costs take the form of volume discounts (e.g., brokers charging smaller commissions for larger trades), then decreasing marginal transaction costs may cause the equilibrium allocation achieved through market activity to depend on the initial allocation (Stavins 1995). This removes a key political advantage of cap-and-trade: the ability to make initial allocation decisions without affecting the equilibrium outcome (Hahn and Stavins 2012). However, market design can minimize transaction costs through, for example, provision of public clearinghouses for trading (Schmalensee and Stavins 2013), and evidence from implemented programs indicates that these costs have generally not been significant (Schmalensee and Stavins 2017b).

[5]Clearly, this is not an issue with carbon taxes.

Performance in the Presence of Uncertainty

Significant uncertainty characterizes the benefits and costs of environmental protection. In the two decades following the publication of Weitzman's (1974) classic "Prices vs. Quantities" article, it was generally acknowledged that cost uncertainty could significantly affect which policy instrument was more efficient for addressing a given externality because the preferred instrument would depend on the relative slopes of the marginal benefit (damage) and marginal cost (abatement) functions. In particular, Weitzman (1974) showed that in the presence of cost uncertainty, a pure price instrument (such as a tax) is more efficient than a quantity instrument (such as cap-and-trade) when (the absolute value of) the slope of the marginal cost function is greater than (the absolute value of) the slope of the marginal benefit function. This is because there is a smaller deadweight loss due to mistaken predictions of future costs. Conversely, when the slope of the marginal benefit function is greater than the slope of the marginal cost function, cap-and-trade is more efficient.

In Weitzman's (1974) analysis, benefit uncertainty on its own has no effect on the relative efficiency of the two instruments because in neither case is the abatement response affected by abatement benefits. However, building on Weitzman (1974), Stavins (1996) found that when there is simultaneous and positively correlated uncertainty about both marginal benefits and marginal costs, it is more likely that a quantity instrument will be more efficient than would be the case in the standard relative slopes analysis. The opposite result will occur if there is negative correlation between benefit and cost uncertainty (i.e., a price instrument is more likely to be more efficient than in the standard analysis).

This finding regarding correlated benefit and cost uncertainty (Stavins 1996) has only recently been applied in the specific context of climate policy (Karp and Traeger 2018), although the original Weitzman (1974) analysis has been used to examine the merits of different carbon pricing strategies since the early 2000s (Hoel and Karp 2002; Newell and Pizer 2003). Newell and Pizer (2003) argued that because GHGs accumulate in the atmosphere (CO_2 can remain in the atmosphere for hundreds of years), changes in emissions at a specific point in time do not significantly alter atmospheric concentrations. Moreover, because climate change is a function of the atmospheric stock of GHGs, the marginal damage function (for any period) is relatively flat. This means that the marginal benefit function is also relatively flat. In contrast to damages, the costs to abate emissions are a function of current policies. This implies that current period marginal costs will have a steeper slope (in absolute value) than current period marginal benefits and that the more efficient policy will be a price instrument (Newell and Pizer 2003).[6]

With a stock pollutant like CO_2, the marginal benefit of reducing emissions in the current period equals the present discounted value of the stream of reductions in current and future marginal damages—also known as the social cost of carbon (Interagency Working Group on Social Cost of Carbon 2016; Aldy et al. 2021). Karp and Traeger (2018) compare the slopes of the current marginal cost and marginal benefits curves (which indicate the avoided social cost of carbon) for a current unit of CO_2 reduction and employ Stavins's (1996) findings regarding the relative efficiency of price and quantity instruments in the presence of simultaneous and correlated uncertainty about benefits and costs.

[6]Using a different analytical model, Hoel and Karp (2002) independently reached the same conclusion regarding stock externalities.

Technological change is a key source of uncertainty. By reducing both current and future abatement costs (due to the persistence of technology effects), technological change creates a positive correlation between uncertainty about abatement costs and (stock-related) damages. This is because future policy makers will take lower abatement costs into account and adopt more stringent emissions targets and thereby reduce the expected future stock of GHGs in the atmosphere. Thus, Karp and Traeger (2018) find a positive correlation between uncertain benefits and uncertain costs that would—as shown by Stavins (1996)—favor a quantity instrument.

It is important to note that Karp and Traeger's (2018) theoretical analysis does not unambiguously support either a quantity or price instrument. Rather, what Newell and Pizer (2003) found to be an unambiguous result for a stock externality becomes ambiguous; that is, it becomes an empirical question, including the effect of correlated uncertainty (Stavins 1996), rather than being solely a question of the relative slopes of the marginal cost versus marginal damages curves.

When they adopt empirical estimates of key parameters from Nordhaus's (2013) DICE model, Karp and Traeger (2018) find that in many cases, a quantity instrument is superior to a price instrument. They conclude that at a minimum, in the presence of uncertainty, the case for a carbon tax over cap-and-trade "is much weaker than widely believed" (Karp and Traeger 2018, 24).[7] In other words, the clear ranking of price over quantity instruments for the stock externality of climate change no longer holds. Given that the instruments may differ in their performance in the presence of uncertainty but that one instrument is not clearly and consistently superior to the other, I include performance in the presence of uncertainty as an area of difference but not significant difference.

Linkage with Other Jurisdictions

Linking policies across jurisdictions such that emission reductions can be redistributed across jurisdictions facilitates cost savings by allowing firms to take advantage of lower-cost abatement opportunities in other jurisdictions. In the international domain, linkage can allow for distributional equity among nations under existing international climate agreements (United Nations 2015) without sacrificing cost-effectiveness (Bodansky et al. 2015; Mehling, Metcalf, and Stavins 2018).

Cap-and-trade systems generate a natural unit of exchange for linkage: allowances, typically denominated in units of carbon content (for fossil fuels) or CO_2 emissions (Jaffe, Ranson, and Stavins 2010). Linkage is also possible between heterogeneous instruments, such as a carbon tax in one jurisdiction and a performance standard in others (Metcalf and Weisbach 2012), although this is more challenging to implement than linkage between two cap-and-trade regimes (Bodansky et al. 2015; Mehling, Metcalf, and Stavins 2018). Furthermore, either a carbon tax or a cap-and-trade system can be linked with policies in other jurisdictions via a carbon border adjustment mechanism (CBAM).[8]

[7] In a separate but related analysis, Pizer and Prest (2019) find that the comparative advantage of one instrument over the other depends more on firms' information and expectations about future policy than on the relative slopes of their marginal benefit and marginal cost curves.

[8] For example, in 2023, the European Union plans to link a CBAM to the allowance price in its Emissions Trading System (ETS).

Effects on Carbon Price Volatility

Under a cap-and-trade mechanism, carbon prices are endogenous (and hence subject to fluctuation and volatility), while under a carbon tax, emissions are endogenous (Goulder and Schein 2013). This means that policies that provide greater certainty about carbon prices and hence mitigation cost, such as taxes or cap-and-trade systems with a “price collar” (i.e., an upper and lower limit on allowance prices), provide less certainty about future emissions (Aldy and Stavins 2012). Typically, the price collar combines an auction price floor with a price that triggers the availability of additional allowances from a reserve. On the other hand, a carbon tax that is vulnerable to change for political reasons may offer less predictability than cap-and-trade in terms of the long-term price signal (Rabe 2018). Of course, the “cap” in a cap-and-trade system may also be subject to change for political reasons.

Turning to short-term price volatility, price movements in a cap-and-trade system can discourage capital investment (Pindyck 2017; Metcalf 2019b), which may undermine political support and reduce investment in new technologies and research and development (Aldy and Stavins 2012). From an economic perspective, it makes sense to allow emissions of a stock pollutant to vary from year to year with changes in economic conditions that affect aggregate abatement costs. This occurs automatically with a carbon tax, but with cap-and-trade, temporal flexibility must be incorporated through provisions for allowance banking (and possibly borrowing). In effect, this redefines the cap as a limit on cumulative emissions over a period of years, rather than a limit on annual emissions (Aldy and Stavins 2012). Conversely, a tax can be designed to automatically adjust to meet some quantity target (Hafstead, Metcalf, and Williams 2017; Metcalf 2019b).

Interactions with Complementary Policies

In jurisdictions where carbon taxes or cap-and-trade systems have actually been implemented, they have invariably been part of a larger set of climate policies. Strong economic arguments can be made for complementing a carbon pricing regime with additional policies when other market failures are present; such market failures include the principal-agent problem that affects incentives for energy efficiency improvements in renter-occupied buildings and information spillovers from research and development activities (Schmalensee and Stavins 2017b). But when these types of market failures do not exist (and if the complementary policies target CO_2 sources that are covered by a carbon pricing regime), problems can arise, as well as differences in the consequences of a carbon tax versus a cap-and-trade system.

Under cap-and-trade, a threefold problem can arise when a complementary policy targets sources that fall under the cap (Schmalensee and Stavins 2017b). First, if the complementary policy is binding, then it achieves no additional emission reductions beyond what is achieved by the cap-and-trade system; instead, emissions simply shift to other sectors under the overall cap (100 percent leakage), unless the complementary policy renders the allowance price floor binding. Second, marginal abatement costs are no longer equalized across sources, which means that aggregate abatement costs are higher than without the additional policy. Third, allowance prices are suppressed, which may diminish incentives for technological change. Thus, the overall effect of the complementary policy is to increase abatement costs and lower allowance prices without further reducing emissions (Goulder and Stavins 2011).

By contrast, pairing a carbon tax with a complementary policy neither creates emissions leakage nor suppresses allowance prices. In fact, the complementary policy can serve to reduce emissions below the level that would be achieved by the tax alone. Nevertheless, combining a tax with other policies affecting the same sources will not be cost-effective because marginal abatement costs will not be identical across all sources. It would be cheaper to achieve the same aggregate emissions reduction by abandoning the complementary policy and increasing the carbon tax rate.

Potential for Corruption and Market Manipulation

Any policy can create the potential for corruption and attempts by firms to manipulate the market. Indeed, although the EU ETS has experienced just one case of allowance theft (in the Czech Republic in 2011), it was quite significant, and allowance accounts have been hacked in a number of countries (Metcalf 2019b). Thus, cap-and-trade systems require regulatory oversight.

With a carbon tax, the parallel concern is tax evasion, which is a significant problem in many countries. In the US context, however, tax evasion is infrequent and does not present major challenges (Metcalf 2019b). In contrast, market manipulation in cap-and-trade systems could be a problem in the United States, although only one case of fraudulent activity has been reported, and it was successfully prosecuted by the US Department of Justice.[9] No cases of fraud have been reported in California's large and ambitious cap-and-trade system or in the northeastern states' electric sector trading program (Schmalensee and Stavins 2017b). That said, it is difficult to contest the conclusion that, at least in the United States (and probably other industrialized countries), "the risk of cybertheft from electronic registries in a cap-and-trade system is likely to present a greater problem than the risk of tax evasion in a carbon tax" (Metcalf 2019b, 82).

Complexity and Administrative Requirements

The simplest cap-and-trade system is likely to involve greater design complexity than the simplest carbon tax, and greater complexity can mean higher administrative costs. At a minimum, under a cap-and-trade system, the regulatory authority must track allowances (at the end of compliance periods), possibly hold auctions, and develop necessary rules. In contrast, a carbon tax can be implemented using existing methods for monitoring and reporting fuel supplies. Thus, administrative costs are likely to be higher for cap-and-trade than for a carbon tax (Goulder and Schein 2013).

However, it is important to note two caveats. First, the actual (marginal) administrative costs of cap-and-trade systems have not been significant (Schmalensee and Stavins 2017b), although the fixed costs of establishing trading institutions are presumably greater than the fixed costs of establishing a carbon tax, especially where the latter can build on existing fossil fuel excise taxes. Second, experience suggests that simple tax proposals can become considerably more complex as they are passed into law. It is not clear whether a policy as important as a national carbon tax would turn out to be "simple" in both its design and implementation.

[9]This occurred in California's Regional Clean Air Incentives Market (RECLAIM).

Hybrid Policy Instruments and a Policy Continuum

An important theme that emerges from the discussion thus far is that some differences between carbon taxes and cap-and-trade diminish with implementation. Indeed, some potential design elements of taxes and cap-and-trade systems result in what could be described as hybrids of price and quantity approaches. For example, auctioning allowances and adding mechanisms such as a price cap or price collar in order to reduce short-term price volatility and/or long-term price uncertainty can make a cap-and-trade system more like a tax (Roberts and Spence 1976). Of course, including a ceiling on allowance prices eliminates an attribute of cap-and-trade that is important to environmental advocates—namely, certainty regarding future emissions (Goulder and Schein 2013). However, any additional allowances the government offers for sale to contain costs could come from an allowance reserve set aside for this purpose or could be borrowed from future allocations, which means that there would be no change in the total quantity of emissions allowed over time (Stavins 2008).

Similarly, a carbon tax can be designed to include features that make it more like cap-and-trade. To provide greater emissions certainty, for example, policy makers could employ various strategies (Aldy et al. 2017). Such strategies could include a formula for adjusting the tax (Hafstead, Metcalf, and Williams 2017), requiring periodic government review (Aldy 2018), or dedicating some tax revenue to mitigation activities (Murray, Pizer, and Reichert 2017). Such features can turn a pure carbon tax into a hybrid instrument, just as adding a price collar can be viewed as turning a pure cap-and-trade system into a hybrid instrument (Borenstein et al. 2019).

This suggests that the two carbon pricing instruments—taxes and cap-and-trade—do not present a dichotomous policy choice. Rather, each presents a choice of design elements along a policy continuum (Weisbach 2010). This means that ultimately, the design details themselves are highly consequential, regardless of which of the two instruments is used; indeed, these details can actually be more important than the choice between the two instruments (Stavins 1997; Keohane, Revesz, and Stavins 1998; Goulder and Hafstead 2018).

Lessons Learned from Experience with Carbon Pricing Policies

As noted in the introduction, more than 60 carbon pricing policies have been implemented or are scheduled for implementation worldwide. In this section, I discuss a few prominent examples that offer the most useful lessons for policy makers considering and possibly designing carbon pricing systems.

Lessons from Experience with Cap-and-Trade Policies

Major emissions trading programs in the United States have included the leaded gasoline phasedown (1982–1987), the sulfur dioxide (SO_2) allowance trading system (1994–2010), a trading program for nitrogen oxides (NO_x; 1998–2009), and RECLAIM in California (1993–present). All of these have been true cap-and-trade programs, except for the leaded gasoline phasedown, which involved a tradable performance standard. In addition, two prominent US trading systems specifically address CO_2 emissions: the Regional Greenhouse Gas Initiative (RGGI), implemented by a group of northeastern states (2009–present), and California's

AB-32/398 system (2013–present). The world's largest carbon pricing initiative is the EU ETS (2008–present). Details on the performance of all of these systems can be found in Schmalensee and Stavins (2017a, 2017b) and Haites et al. (2018).

Environmental and economic performance

Programs such as the US lead phasedown and SO_2 allowance trading have demonstrated that cap-and-trade can be environmentally effective and economically cost-effective. For example, SO_2 trading is estimated to have reduced aggregate abatement costs by more than half relative to a well-designed command and control approach (Metcalf 2019b). RGGI has been credited with achieving roughly half of the region's CO_2 reductions observed since the program was launched, with the remaining reductions attributed to low natural gas prices, economic recession, and other policies (Murray and Maniloff 2015). The EU ETS had a number of problems in its pilot phase (Metcalf 2019b) but has since functioned as anticipated (Ellerman, Convery, and Perthuis 2010), producing significant emission reductions (Bayer and Aklin 2020) and inducing climate-friendly technological change (Calel 2020), with no observable negative impact on the economic performance of regulated firms (Dechezleprêtre, Nachtigall, and Venmans 2018).

To date, the scope of coverage in most CO_2 trading programs has been limited. This is due to the common use of downstream sector-specific programs (e.g., that target the electricity sector), although economy-wide systems have been shown to be feasible in California. This downstream sector-specific focus contrasts with the textbook approach of trading carbon rights upstream based on fuel carbon content (World Bank Group 2016).

Transaction costs, monitoring, and enforcement

Overall, transaction costs have been small to trivial, particularly when compliance entities are homogeneous, as in the lead phasedown and SO_2 trading programs. Experience confirms the importance of establishing a cap that is well below business as usual emissions to ensure a robust allowance market (SO_2 trading, RECLAIM). Indeed, other government regulations and judicial decisions eventually eliminated the SO_2 market (Schmalensee and Stavins 2013). High levels of compliance require a combination of strict monitoring, reporting, and verification and significant penalties for noncompliance (SO_2 trading). In addition, setting final rules well before the first compliance period can avoid unnecessary price volatility (SO_2 trading, NO_x trading, EU ETS).

Experience shows that simplicity and transparency are also important. For example, transparent rules can make allowance allocations difficult to contest or manipulate, particularly if rules are clearly defined up front. Avoiding requirements for prior regulatory approval of individual trades can also help limit uncertainty and reduce transaction costs (lead phasedown, SO_2 trading).

Flexibility mechanisms

Experience also indicates the importance of flexible systems that allow for a broad set of compliance alternatives, in terms of timing and technology options. Studies have shown that under less flexible systems, market-based instruments would not have induced as much technological change (Ellerman and Montero 1998; Keohane 2003; Schmalensee and Stavins 2013) or process innovation (Doucet and Strauss 1994).

Provisions for allowance banking have been important and account for a large share of realized gains from trade (lead phasedown, SO_2 trading). In contrast, the absence of banking provisions can lead to allowance price spikes (RECLAIM) and collapses (EU ETS). More broadly, a changing economy can render a cap nonbinding (RGGI, EU ETS) or drive allowance prices to excessive levels (RECLAIM). Thus, there is a key role for mechanisms that establish both a floor and a ceiling on allowance prices (RGGI, AB-32/398). A comparison of such price collars (used in RGGI and AB-32/398) with quantity collars (modeled after the EU ETS Market Stability Reserve)[10] found that price collars were superior to quantity collars in terms of reducing price volatility and increasing efficiency (Holt and Shobe 2016). However, excessive constraints on the use of offsets can render this mechanism ineffective for cost containment (RGGI, AB-32/398).

Allowance allocations

Free allowance allocation has been used to gain initial political support for trading programs, although this strategy forgoes the opportunity to reduce overall social cost by using revenues from auctions to cut distortionary taxes (SO_2 trading, AB-32/398). Where allowances have been auctioned, the revenues have more often been used to fund new or existing government programs, not to cut distortionary taxes as economists would recommend (AB-32/398, RGGI). In fact, allowance auctions have generated very significant revenue for governments (RGGI, AB-32/398).

Over time, there has been a trend toward greater use of allowance auctioning in cap-and-trade systems (AB-32/398, RGGI, EU ETS), which has resulted in more systems that can be characterized as hybrids of price and quantity instruments. Other design elements aimed at containing costs or reducing short-term price volatility and/or long-term price uncertainty have further blurred the distinction between cap-and-trade systems and a tax.

The increasing use of price collars has reinforced the trend toward hybrid systems. Although the price ceiling in the California cap-and-trade system has been "soft" in the sense that the reserve supply was finite, the state's post-2020 system includes a hard cap on prices by making an unlimited number of allowances available at a specified, escalating price level (Schatzki and Stavins 2018), further reinforcing the trend toward truly hybrid systems.

Competitiveness impacts

The competitiveness impacts of cap-and-trade systems have been a prominent political concern. In practice, leakage from cap-and-trade systems has ranged from nonexistent (lead phasedown) to potentially serious, as in the case of RGGI (Fell and Maniloff 2018). Free allowance allocation will not address these problems per se (EU ETS), but an output-based updating allowance allocation can provide an effective response (AB-32/398).

[10]The Market Stability Reserve adjusts the number of allowances available at auction.

Complementary policies

Although complementary policies can play a constructive role in a carbon pricing regime, in practice, such policies have frequently conflicted with trading programs, causing emissions to be relocated rather than reduced, driving up abatement costs, and suppressing allowance prices (Organization for Economic Cooperation and Development 2011). This perverse outcome has occurred in two of the most prominent applications of cap-and-trade for climate mitigation purposes (AB-32/398, EU ETS).

The extent of such perverse interactions between a complementary policy and the cap-and-trade system within which it is nested can be measured by the difference in marginal abatement costs between the two policies. For example, in spring 2021, allowances in California's two distinct trading systems—the AB-32/398 program and the state's refinery-based Low Carbon Fuel Standard—were trading at about $19 per ton of CO_2 and nearly $200 per ton, respectively. This very large gap indicates a dramatic departure from cost-effective policy design. Such perverse policy interactions (Fankhauser, Hepburn, and Park 2010) may be partially addressed through reforms such as the market stability reserves introduced in the EU, California, and RGGI trading systems (Perino, Ritz, and van Benthem 2019).

Lessons from Experience with Carbon Taxes

In contrast to cap-and-trade, most pollution taxes, including the 30 carbon taxes (as of 2020) identified by the World Bank (World Bank Group 2020), thus far have focused on CO_2 (or closely related energy generation). The high carbon taxes adopted by several northern European countries and a more recent carbon tax implemented in British Columbia are the most prominent applications of this instrument.

Northern European carbon taxes

In the 1990s, several northern European countries implemented carbon taxes as part of broader tax policy changes, frequently as excise tax reforms (Murray and Rivers 2015). These policies have differed in the scope of their coverage and their tax rates, and many have been in place simultaneously (or even linked) with the EU ETS, making it difficult to assess their impacts (Murray and Rivers 2015). Some of these countries, such as Sweden, now impose the highest carbon prices in the world, although there are significant variations in the effective tax per unit of CO_2 across fuels and industries within each country (Bruvoll and Larsen 2004; Daugjberg and Pedersen 2004). Provisions to cushion fiscal impacts have been common, especially where industries have raised concerns about international competitiveness (Aldy and Stavins 2012). As a result, carbon taxes have typically not provided uniform price signals across emission sources.

Unfortunately, there is only limited empirical evidence on the emissions impacts of these carbon taxes. In 1991, Sweden implemented a CO_2 tax of $33 per ton as a part of fiscal reforms that cut income tax rates (Speck 2008). The tax, which has since increased to about $120 per ton, has generated considerable revenue for Sweden's general budget, as the government does not earmark tax revenues (Government Offices of Sweden 2019). One recent analysis indicates that transport sector CO_2 emissions fell after the country implemented both carbon and value-added taxes on transport fuel (Andersson 2019).

In 1991, Norway implemented a carbon tax that applied different rates to different sectors. In 2009, the tax covered about 55 percent of the country's GHG emissions, while a trading scheme linked to the EU ETS covered an additional 13 percent of emissions. Denmark's tax began at about $18 per ton of CO_2 in 1992 and reached $28 per ton by 2020, but with a lower rate for manufacturers (Speck 2008; World Bank Group 2020). Finland has imposed a general tax on energy, coupled with a carbon-based fuel surtax, since 1997. The tax on CO_2 ranges from about $58 to $68 per ton, depending on the sector (World Bank Group 2020).

British Columbia's carbon tax

Established in 2008, British Columbia's carbon tax comes closest to the "ideal" form recommended by economists (Metcalf 2019b). The tax, which is part of a broader plan to cut provincial GHG emissions by a third (British Columbia 2021), was intended to be economy-wide, covering approximately 70–75 percent of GHG emissions. It is collected upstream at the wholesale level (from fuel distributors) on the basis of fuel carbon content (Duff 2008). The tax does not apply to non–fossil fuel GHG emissions (including from industrial processes, landfills, forestry, and agriculture) or methane. In addition, fuel exports, including virtually all of the coal mined in the province (Murray and Rivers 2015), fuels used by aviation and shipping into or out of the province, and operations and fuels used in agriculture (since 2012), are exempt.

British Columbia's tax began at $7.50 per ton of CO_2 in 2008 and increased gradually thereafter; it is now $32 per ton and is scheduled to increase to $40 in 2022 (British Columbia 2021). Originally, all of the revenue was to be refunded through tax cuts to businesses and individuals, with low-income individuals protected through a targeted tax credit. Over time, the policy has evolved to provide additional tax cuts to specific sectors and locations (Murray and Rivers 2015; Goulder and Hafstead 2018). Considering both use-side and source-side impacts, the tax appears to be progressive, even without accounting for revenue use (Beck et al. 2015). However, empirical evidence on the policy's environmental impacts is unclear. One study estimates province-wide CO_2 reductions of 5–10 percent, with little negative economic impact (Metcalf 2019b) but with unknown emissions leakage (Murray and Rivers 2015). Other research questions whether emissions have been reduced at all (Pretis 2019).

Summary and Conclusions

This article has identified the theoretical and empirical similarities and differences between carbon taxes and cap-and-trade, examined their advantages and disadvantages, and highlighted some of the key lessons learned from experience with these policies. I find that when carbon taxes and carbon cap-and-trade systems are designed in ways that make them truly comparable, their characteristics and outcomes are similar and, in some respects, fully equivalent, particularly concerning key dimensions such as emission reductions, abatement costs, possibilities for raising revenue, costs to regulated firms, distributional impacts, and effects on competitiveness. However, the two approaches can perform quite differently in terms of other dimensions, sometimes favoring taxes and sometimes favoring cap-and-trade. In terms of complexity and administrative requirements, interaction with complementary policies, and effects on carbon price volatility, a tax has clear advantages. On the other hand, cap-and-trade may have advantages in terms of ease of linkage with policies in other jurisdictions and, possibly, anticipated performance in the presence of uncertainty. The major differences between the two approaches depend on the details of program design. Indeed, I argue that what appears at first glance to be a dichotomous choice between two distinct instruments can end up being a choice of specific design elements along a policy continuum.

I find that neither approach clearly dominates in terms of anticipated performance, and any ranking of the two instruments in a specific context would likely depend on the weight given to different policy considerations (Goulder and Hafstead 2018). When there are relatively minor differences in impacts, a key question becomes which of the two approaches is more politically feasible and which is more likely to be well designed (Furman et al. 2007). Judgments about political feasibility might be informed by the revealed preference of jurisdictions that have adopted carbon pricing strategies, but thus far carbon tax and CO_2 cap-and-trade systems have been introduced in roughly equal numbers around the world (World Bank Group 2020). A more informative comparison of the two approaches might be to focus instead on the combination of the scope of coverage and program stringency (measured by allowance price or tax level), in which case cap-and-trade stands out as the more important approach worldwide, at least for the present time.

An examination of which type of system is more likely to be better designed is consistent with the notion that a policy's net benefits should be weighted by the probability that the policy will be implemented in the first place (Pizer, Stavins, and Stuart 2020). Indeed, even if carbon pricing is the first-best policy for the long term, other strategies may be more politically achievable in the short term (Gillis 2018; Goulder 2019). To illustrate this point, while 61 jurisdictions have carbon taxes or cap-and-trade policies, fully 176 countries have adopted renewable energy policies or energy efficiency standards, and another 110 jurisdictions have implemented feed-in tariffs (Carattini, Carvalho, and Fankhauser 2017). This highlights that building political acceptance for pricing instruments more broadly could be a key step toward more effective climate policy action.

Public perceptions—some of them inaccurate—appear to be a primary reason for aversion to carbon pricing (Carattini, Carvalho, and Fankhauser 2017). Hence, one approach might be to improve public understanding of such policies through better media coverage, not to mention broader educational initiatives. But careful policy design would be another and perhaps more effective approach. For example, departing from first-best design may make carbon emission taxes or caps more palatable (Jaccard 2012), as suggested by the gradual phased approach in California and British Columbia. Of course, this could lead to a carbon price becoming stuck at a level that is too low to achieve the environmental objective, in which case a commitment device, such as a trajectory of tax levels or caps over time, may be helpful. For example, the legislation that established the US SO_2 trading program and the British Columbia and Swedish carbon taxes included such trajectories for future caps and tax rates.

Another departure from first-best design would be to earmark revenues to finance additional mitigation (Kotchen, Turk, and Leiserowitz 2017) or to address equity concerns. Either pricing instrument can be designed to be revenue neutral by including lump-sum rebates of tax (or auction) revenue or by using revenue to cushion impacts on low-income or other burdened constituencies (Amdur, Rabe, and Borick 2014; Carattini, Carvalho, and Fankhauser 2017). But as surveys in several countries suggest, there is a distinct lack of public support for using revenue to cut distortionary taxes (Carattini, Carvalho, and Fankhauser 2017). Perhaps voters do not understand the logic of the "double dividend" (Goulder 1995), or perhaps they simply harbor doubts that governments will actually follow through on cutting other taxes. In contrast, in many countries, there is support for using carbon pricing revenue to ease impacts on low-income households (Carattini, Carvalho, and Fankhauser 2017). In fact, proposals from across the US political spectrum have featured carbon dividend or cap-and-dividend systems, which return revenues directly to citizens (Boyce and Riddle 2007; Sedor 2015; US House of Representatives 2018; Akerlof et al. 2019).

Is there now a political opening for well-designed carbon taxes in the United States? The successful demonization of the Obama administration's CO_2 cap-and-trade proposal as "cap-and-tax" (*Wall Street Journal* 2009) might present such an opening for more serious consideration of a carbon tax in this country. In particular, large budget deficits may heighten interest in new sources of revenue, with consumption taxes—such as energy or carbon taxes—likely to be particularly attractive in this regard. The past few years have witnessed support for such strategies, not only in the broader policy community and among the usual Democratic sources of support but also among prominent Republican academic economists and former high-level government officials (Akerlof et al. 2019).

However, the prospects for support among current elected US officials are decidedly less clear. Indeed, if it was possible to rally opposition to cap-and-trade by calling it "cap-and-tax," it will be that much easier to label a carbon tax a "tax." Even the progressive politicians who authored the Green New Deal in the United States did not make a carbon tax central to their proposal. Given that any national carbon pricing policy appears likely to continue to face an uphill battle in the United States, it may be more productive for economists to redirect some of their attention to designing better second-best nonpricing policies, such as better performance standards. At some point, however, the politics surrounding climate policy will change. Thus, it is important that research on carbon pricing instruments continues, particularly if it is focused on policies that are likely (sooner or later) to be politically feasible, whether in the United States or other countries.

References

Akerlof, G., R. Aumann, A. Deaton, P. Diamond, R. Engle, E. Fama, L. P. Hansen, et al. 2019. Economists' statement on carbon dividends. *Wall Street Journal*, January 16.

Aldy, J. E. 2018. Carbon tax review and updating. Unpublished paper, John F. Kennedy School of Government, Harvard University, Cambridge, MA.

Aldy, J. E., M. Hafstead, G. E. Metcalf, B. C. Murray, W. A. Pizer, C. Reichert, and R. W. Williams. 2017. Resolving inherent uncertainty of carbon taxes. *Harvard Environmental Law Review* 41: 1–13.

Aldy, J. E., M. J. Kotchen, R. N. Stavins, and J. H. Stock. 2021. Keep climate policy focused on the social cost of carbon. *Science* 373 (6557): 850–52.

Aldy, J. E., and W. A. Pizer. 2015. Competitiveness impacts of climate change mitigation policies. *Journal of the Association of Environmental and Resource Economists* 2 (4): 565–95.

Aldy, J. E., and R. N. Stavins. 2012. The promise and problems of pricing carbon: Theory and practice. *Journal of Environment and Development* 21 (2): 152–80.

Amdur, D., B. G. Rabe, and C. Borick. 2014. Public views on a carbon tax depend on the proposed use of revenue. *Issues in Energy and Environmental Policy* 13: 1–9.

Andersson, J. J. 2019. Carbon taxes and CO_2 emissions: Sweden as a case study. *American Economic Journal* 11 (4): 1–30.

Bayer, P., and M. Aklin. 2020. The European Union Emissions Trading System reduced CO_2 emissions despite low prices. *Proceedings of the National Academy of Sciences of the USA* 117 (16): 8804–12.

Beck, M., N. Rivers, R. Wigle, and H. Yonezawa. 2015. Carbon tax and revenue recycling: Impacts on households in British Columbia. *Resource and Energy Economics* 41: 40–69.

Bodansky, D., S. Hoedl, G. Metcalf, and R. Stavins. 2015. Facilitating linkage of climate policies through the Paris outcome. *Climate Policy* 16 (8): 956–72.

Borenstein, S., J. Bushnell, F. A. Wolak, and M. Zaragoza-Watkins. 2019. Expecting the unexpected: Emissions uncertainty and environmental market design. *American Economic Review* 109 (11): 3953–77.

Bovenberg, A. L., and L. H. Goulder. 1996. Optimal environmental taxation in the presence of other taxes: General equilibrium analyses. *American Economic Review* 86 (4): 985–1000.

Boyce, J. K., and M. Riddle. 2007. Cap and dividend: How to cap global warming while protecting the incomes of American families. Working Paper 150, Political Economy Research Institute, University of Massachusetts, Amherst.

British Columbia. 2021. British Columbia's carbon tax. https://www2.gov.bc.ca/gov/content/environment/climate-change/clean-economy/carbon-tax.

Bruvoll, A., and B. M. Larsen. 2004. Greenhouse gas emissions in Norway: Do carbon taxes work? *Energy Policy* 32: 493–505.

Bushnell, J., and Y. Chen. 2012. Allocation and leakage in regional cap-and-trade markets for CO_2. *Resource and Energy Economics* 34: 647–68.

Calel, R. 2020. Adopt or innovate: Understanding technological responses to cap-and-trade. *American Economics Journal* 12 (3): 170–201.

Carattini, S., M. Carvalho, and S. Fankhauser. 2017. How to make carbon taxes more acceptable. Policy report, Grantham Research Institute on Climate Change and the Environment and Centre for Climate Change Economics and Policy, London School of Economics and Political Science, London.

Carl, J., and D. Fedor. 2016. Tracking global carbon revenues: A survey of carbon taxes versus cap-and-trade in the real world. *Energy Policy* 96: 50–77.

Cosbey, A., S. Droege, C. Fischer, and C. Munnings. 2019. Developing guidance for implementing border carbon adjustments: Lessons, cautions, and research needs from the literature. *Review of Environmental Economics and Policy* 13 (1): 3–22.

Daugjberg, C., and A. B. Pedersen. 2004. New policy ideas and old policy networks: Implementing green taxation in Scandinavia. *Journal of Public Policy* 24 (2): 219–49.

Dechezleprêtre, A., D. Nachtigall, and F. Venmans. 2018. The joint impact of the European Union Emissions Trading System on carbon emissions and economic performance. OECD Economics Department Working Papers no. 1515, Organization for Economic Cooperation and Development, Paris.

Doucet, J. A., and T. Strauss. 1994. On the bundling of coal and sulfur dioxide emissions allowances. *Energy Policy* 22 (9): 764–70.

Duff, D. G. 2008. Carbon taxation in British Columbia. *Vermont Journal of Environmental Law* 10: 85–105.

Ellerman, A. D., F. J. Convery, and C. de Perthuis, eds. 2010. *Pricing carbon: The European Union emissions trading scheme*. Cambridge: Cambridge University Press.

Ellerman, A. D., P. L. Joskow, and D. Harrison. 2003. *Emissions trading in the U.S.: Experience, lessons, and considerations for greenhouse gases*. Arlington, VA: Pew Center on Global Climate Change.

Ellerman, A. D., and J.-P. Montero. 1998. The declining trend in sulfur dioxide emissions: Implications for allowance prices. *Journal of Environmental Economics and Management* 36: 26–45.

Fankhauser, S., C. Hepburn, and J. Park. 2010. Combining multiple climate policy instruments: How not to do it. *Climate Change Economics* 1 (3): 209–25.

Fell, H., and P. Maniloff. 2018. Leakage in regional environmental policy: The case of the regional greenhouse gas initiative. *Journal of Environmental Economics and Management* 87: 1–23.

Flannery, B., J. Hillman, J. Mares, and M. Porterfield. 2018. Framework proposal for a US upstream greenhouse gas tax with WTO-compliant border adjustments. RFF report, Resources for the Future, Washington, DC, October.

Furman, J., J. E. Bordoff, M. Deshpande, and P. J. Noel. 2007. An economic strategy to address climate change and promote energy security. Strategy paper, Hamilton Project, Brookings Institution, Washington, DC.

Gillis, J. 2018. Forget the carbon tax for now. *New York Times*, December 27.

Goulder, L. H. 1995. Environmental taxation and the "double dividend": A reader's guide. *International Tax and Public Finance* 2 (2): 277–313.

———. 2019. Addressing the urgency of climate-change policy. Working paper, Stanford University, Stanford, CA, March.

Goulder, L. H., and M. A. C. Hafstead. 2018. *Confronting the climate challenge: U.S. policy options*. New York: Columbia University Press.

Goulder, L. H., M. A. C. Hafstead, G. Kim, and X. Long. 2018. Impacts of a carbon tax across U.S. household income groups: What are the equity-efficiency trade-offs? NBER Working Paper no. 25181, National Bureau of Economic Research, Cambridge, MA, October.

Goulder, L. H., and A. R. Schein. 2013. Carbon taxes versus cap and trade: A critical review. *Climate Change Economics* 4 (3): 1–28.

Goulder, L. H., and R. N. Stavins. 2011. Challenges from state-federal interactions in U.S. climate change policy. *American Economic Review Papers and Proceedings* 101 (3): 253–57.

Government Offices of Sweden. 2019. Sweden's carbon tax. https://www.government.se/government-policy/swedens-carbon-tax/.

Hafstead, M., G. E. Metcalf, and R. C. Williams. 2017. Adding quantity certainty to a carbon tax: The role of a tax adjustment mechanism for policy pre-commitment. *Harvard Environmental Law Review* 41: 41–57.

Hahn, R. W., and R. N. Stavins. 2012. The effect of allowance allocations on cap-and-trade system performance. *Journal of Law and Economics* 54 (4): S267–S294.

Haites, E., D. Maosheng, K. Sims Gallagher, S. Mascher, E. Narassimhan, K. Richards, and M. Wakabayashi. 2018. Experience with carbon taxes and greenhouse gas emissions trading systems. *Duke Environmental Law and Policy Forum* 29: 109–82.

Hoel, M., and L. Karp. 2002. Taxes versus quotas for a stock pollutant. *Resource and Energy Economics* 24: 367–84.

Holt, C. A., and W. M. Shobe. 2016. Price and quantity collars for stabilizing emission allowance prices: Laboratory experiments on the EU ETS market stability reserve. *Journal of Environmental Economics and Management* 76: 32–50.

Interagency Working Group on Social Cost of Carbon. 2016. Technical update of the social cost of carbon for regulatory impact analysis under Executive Order 12866. Technical support document, US government, Washington, DC.

Jaccard, M. 2012. The political acceptability of carbon taxes: Lessons from British Columbia. In *Handbook of research on environmental taxation*, eds. Milne, J. E., and M. S. Andersen, 175–91. Cheltenham, UK: Edward Elgar.

Jaffe, J., M. Ranson, and R. N. Stavins. 2010. Linking tradable permit systems: A key element of emerging international climate policy architecture. *Ecology Law Quarterly* 36: 789–808.

Jung, C., K. Krutilla, and R. Boyd. 1996. Incentives for advanced pollution abatement technology at the industry level: An evaluation of policy instruments. *Journal of Environmental Economics and Management* 30: 95–111.

Kaplow, L. 2010. Taxes, permits, and climate change. NBER Working Paper no. 16268, National Bureau of Economic Research, Cambridge, MA.

Karp, L., and C. Traeger. 2018. Prices versus quantities reassessed. CESifo Working Paper 7331, Munich Society for the Promotion of Economic Research, Munich, November.

Keohane, N. O. 2003. What did the market buy? Cost savings under the U.S. tradeable permits program for sulfur dioxide. Working Paper YCELP-01-11-2003, Yale Center for Environmental Law and Policy, New Haven, CT.

———. 2009. Cap and trade, rehabilitated: Using tradable permits to control U.S. greenhouse gases. *Review of Environmental Economics and Policy* 3: 42–62.

Keohane, N. O., R. L. Revesz, and R. N. Stavins. 1998. The choice of regulatory instruments in environmental policy. *Harvard Environmental Law Review* 22 (2): 313–67.

Knittel, C. R. 2019. Diary of a wimpy carbon tax: Carbon taxes as federal climate policy. CEEPR WP 2019-013, MIT Center for Energy and Environmental Policy Research, Cambridge, MA, August.

Kotchen, M. J., Z. M. Turk, and A. A. Leiserowitz. 2017. Public willingness to pay for a U.S. carbon tax and preferences for spending the revenue. *Environmental Research Letters* 12: 094012.

Mankiw, L. G. 2006. The Pigou Club manifesto. *Greg Mankiw's Blog*. http://gregmankiw.blogspot.com/2006/10/pigou-club-manifesto.html.

Mehling, M. A., G. E. Metcalf, and R. N. Stavins. 2018. Linking climate policies to advance global mitigation. *Science* 359 (6379): 997–98.

Metcalf, G. E. 2007. A proposal for a U.S. carbon tax swap: An equitable tax reform to address global climate change. Discussion Paper 2007-12, Hamilton Project, Brookings Institution, Washington, DC.

———. 2009. Market-based policy options to control U.S. greenhouse gas emissions. *Journal of Economic Perspectives* 23 (2): 5–27.

———. 2019a. The distributional impacts of U.S. energy policy. *Energy Policy* 129 (June 2019): 926–29.

———. 2019b. *Paying for pollution: Why a carbon tax is good for America*. New York: Oxford University Press.

Metcalf, G. E., and D. Weisbach. 2012. Linking policies when tastes differ: Global climate policy in a heterogeneous world. *Review of Environmental Economics and Policy* 6 (1): 110–29.

Milliman, S. R., and R. Prince. 1989. Firm incentives to promote technological change in pollution control. *Journal of Environmental Economics and Management* 17: 247–65.

Murray, B., and P. Maniloff. 2015. Why have greenhouse gas emissions in RGGI states declined? An econometric attribution to economic, energy market, and policy factors. *Energy Economics* 51: 581–89.

Murray, B., W. A. Pizer, and C. Reichert. 2017. Increasing emissions certainty under a carbon tax. *Harvard Environmental Law Review* 41: 14–27.

Murray, B., and N. Rivers. 2015. British Columbia's revenue-neutral carbon tax: A review of the latest "grand experiment" in environmental policy. *Energy Policy* 86: 674–83.

Newell, R. G., A. B. Jaffe, and R. N. Stavins. 1999. The induced innovation hypothesis and energy-saving technological change. *Quarterly Journal of Economics* 114: 941–75.

Newell, R. G., and W. A. Pizer. 2003. Regulating stock externalities under uncertainty. *Journal of Environmental Economics and Management* 45: 416–32.

Newell, R. G., and R. N. Stavins. 2003. Cost heterogeneity and the potential savings from market-based policies. *Journal of Regulatory Economics* 23: 43–59.

Nordhaus, W. D. 2007. To tax or not to tax: Alternative approaches to slowing global warming. *Review of Environmental Economics and Policy* 1: 26–44.

———. 2013. *The climate casino: Risk, uncertainty, and economics for a warming world*. New Haven, CT: Yale University Press.

Organization for Economic Cooperation and Development. 2011. Interactions between emission trading systems and other overlapping policy instruments. General distribution document, Environment Directorate, Organization for Economic Cooperation and Development, Paris.

Parry, I. W. H., R. C. Williams, and L. H. Goulder. 1999. When can carbon abatement policies increase welfare? The fundamental role of distorted factor markets. *Journal of Environmental Economics and Management* 37: 52–84.

Perino, G., R. A. Ritz, and A. van Benthem. 2019. Understanding overlapping policies: Internal carbon leakage and the punctured waterbed. NBER Working Paper no. 25643, National Bureau of Economic Research, Cambridge, MA, March.

Pindyck, R. S. 2017. Coase Lecture—taxes, targets, and the social cost of carbon. *Economica* 84: 345–64.

Pizer, W. A., and B. C. Prest. 2019. Prices versus quantities with policy updating. NBER Working Paper no. 22379, National Bureau of Economic Research, Cambridge, MA, June.

Pizer, W. A., R. N. Stavins, and D. N. Stuart. 2020. Incorporating political feasibility into economic analysis: The expected net benefits of alternative climate change policy instruments. Cambridge, MA: Harvard Environmental Economics Program.

Pretis, F. 2019. Does a carbon tax reduce CO_2 emissions? Evidence from British Columbia. SSRN. https://ssrn.com/abstract=3329512.

Rabe, B. 2018. *Can we price carbon?* Cambridge, MA: MIT Press.

Roberts, M. J., and M. Spence. 1976. Effluent charges and licenses under uncertainty. *Journal of Public Economics* 5: 193–208.

Schatzki, T., and R. N. Stavins. 2018. Key issues facing California's GHG cap-and-trade system for 2021–2030. M-RCBG Faculty Working Paper Series no. 2018-02, Mossavar-Rahmani Center for Business and Government, Harvard Kennedy School, Cambridge, MA.

Schmalensee, R., and R. N. Stavins. 2013. The SO_2 allowance trading system: The ironic history of a grand policy experiment. *Journal of Economic Perspectives* 27 (1): 103–22.

———. 2017a. The design of environmental markets: What have we learned from experience with cap and trade? *Oxford Review of Economic Policy* 33 (4): 572–88.

———. 2017b. Lessons learned from three decades of experience with cap and trade. *Review of Environmental Economics and Policy* 11 (1): 59–79.

Scotchmer, S. 2011. Cap-and-trade, emissions taxes, and innovation. *Innovation Policy and the Economy* 11: 29–54.

Sedor, N. 2015. Why fee and dividend is better than cap and trade at fighting climate change. *Los Angeles Times*, March 5.

Speck, S. 2008. The design of carbon and broad-based energy taxes in European countries. *Vermont Journal of Environmental Law* 10: 31–59.

Stavins, R. N. 1995. Transaction costs and tradeable permits. *Journal of Environmental Economics and Management* 29: 133–46.

———. 1996. Correlated uncertainty and policy instrument choice. *Journal of Environmental Economics and Management* 30: 218–32.

———. 1997. What can we learn from the grand policy experiment? Positive and normative lessons from SO_2 allowance trading. *Journal of Economic Perspectives* 12 (3): 69–88.

———. 2007. A U.S. cap-and-trade system to address global climate change. Discussion Paper 2007-13, Hamilton Project, Brookings Institution, Washington, DC.

———. 2008. A meaningful U.S. cap-and-trade system to address climate change. *Harvard Environmental Law Review* 32 (2): 293–371.

———. 2019. The future of U.S. carbon-pricing policy. NBER Working Paper no. 25912, National Bureau of Economic Research, Cambridge, MA.

United Nations. 2015. Paris agreement. December 12.

US House of Representatives. 2018. Energy innovation and carbon dividend act of 2018. H.R., 7173, November 27.

Wall Street Journal. 2009. The cap and tax fiction. *Wall Street Journal*, June 26.

Weisbach, D. 2010. Instrument choice is instrument design. In *U.S. energy tax policy*, ed. Metcalf, G. E., 113–58. Cambridge: Cambridge University Press.

Weitzman, M. L. 1974. Prices vs. quantities. *Review of Economic Studies* 41 (4): 477–91.

World Bank Group. 2016. *Emissions trading in practice: Handbook on design and implementation*. Washington, DC: Partnership for Market Readiness and International Carbon Action Partnership, World Bank Group.

———. 2020. *State and trends of carbon pricing*. Washington, DC: World Bank Group.