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Chapter Author(s):  Hilary Sigman

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# Monitoring and Enforcement of Climate Policy

Hilary Sigman

Without effective enforcement, public climate policies may not cause changes in private actions. However, the economics literature has not devoted much attention to enforcement of climate policies, with a few notable exceptions (Kruger and Pizer 2004; Johnstone 2005; Kruger and Egenhofer 2006; Silva and Zhu 2008). In part, the inattention to these issues may stem from the view that climate policies will be easy to enforce, at least relative to other air pollution controls. The empirical evidence reviewed in this chapter does support the view that restrictions on carbon dioxide emissions from point sources can be enforced with moderate cost.

However, enforcement of other aspects of climate policy can be daunting. Enforcement is sometimes a dominant consideration in the design of responses to greenhouse gases other than carbon dioxide and to carbon dioxide from sources other than fossil fuels. As a consequence, climate policy poses the novel challenge of integrating easy-to-enforce and difficult-to-enforce components in one policy.

This chapter investigates monitoring and enforcement of climate policy in practice and suggests several lessons from this experience. First, under the European Union (EU) Emission Trading System (ETS), incentives for compliance may derive at least as much from informal costs as official penalties. Second, prices in the EU ETS do not suggest much concern about differential validity of allowances from within the capped sector and offsets

Hilary Sigman is professor of economics at Rutgers University, a nonresident fellow of Resources for the Future, and a research associate of the National Bureau of Economic Research.

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from sources outside this sector. Finally, the empirical evidence points to substantial variation in monitoring and enforcement costs across these different compliance methods.

Given the differences in monitoring and enforcement costs across compliance methods, the government may be tempted to restrict climate policies to areas where enforcement is relatively easy. However, a simple model of enforcement illustrates that expanding markets may not lower compliance. Although deterrence may fall when the government must monitor more complicated activities, a broader market may also lower allowance prices and decrease the incentive to violate the policy. Thus, the effect of expanding the market will depend on the relative strength of these two opposing effects.

## 13.1 Incentives for Compliance with a Climate Policy

In this section, I present a basic model of enforcement of incentive-based environmental policy that has been used extensively in the prior literature (e.g., Harford 1978; Stranlund and Dhanda 1999; Stranlund, Chavez, and Field 2002). The model yields one simple insight that I use to analyze practical enforcement issues in the rest of the chapter.

### 13.1.1 The Compliance Decision

The standard environmental enforcement model considers a risk-neutral emitter who seeks to minimize the sum of compliance cost plus the expected punishment.[1]

Compliance costs depend on the form of the public policy. With a performance standard, compliance costs are just the costs of reducing emissions, $c(e_i, \gamma_i)$, where $e_i$ is the emission level of emitter $i$, and $\gamma_i$ reflects cost heterogeneity across the emitters. An incentive-based policy adds to the compliance cost a term that reflects net outlays (purchases or sales) of allowances or tax paid on emissions. Under a cap-and-trade program, an emitter with initial allowance allocation of $Q_i$ thus has a compliance cost of $c(e_i, \gamma_i) + p \times (q_i - Q_i)$, where $p$ is the equilibrium permit price and $q_i$ the quantity of permits the emitter applies to its own emissions. A carbon tax is similar, but $q_i$ is the level of emissions the emitter reports as its tax base, $p$ is the tax, and $Q_i = 0$. The important implication is that an incentive-based policy gives the emitter choices on two margins, $e_i$ and $q_i$.

The expected penalty depends on the chance a violation is detected, $D(v_i)$, and the fine, $F(v_i)$, each of which is, in general, a function of the magnitude of the violation $v_i$. For either emissions trading or a carbon tax, the violation is $v_i = e_i - q_i$, the difference between actual emissions and $q_i$.

In addition to a fine, most environmental policies require the violator to

1. Polluters may be risk averse, which would tend to strengthen the incentives for compliance, but not fundamentally change the problem. Malik (1990) models emission-market enforcement with risk averse polluters.

"fix" the violation. This requirement attempts to reduce the probability that violating the law is the least-cost option. Emission trading systems often implement this requirement by having violators surrender enough allowances to cover their emissions, perhaps withholding them from the violator's next-year allocation. Thus, the penalty is the fine plus the value of permits surrendered: $F(e_i - q_i) + p \times (e_i - q_i)$ (ignoring discounting if permits are surrendered next year). This penalty is multiplied by the chance the violation is detected, $D(e_i - q_i)$, to form the expected penalty.

Thus, the emitter's problem is to minimize total expected cost subject to the constraint that the violation is nonnegative:

$$(1) \quad \min_{e_i, q_i} \quad c(e_i, \gamma_i) + p \times (q_i - Q_i) + D(e_i - q_i)[F(e_i - q_i) + p \times (e_i - q_i)]$$
$$\text{s.t.} \quad e_i - q_i \geq 0$$

The first-order condition with respect to $q_i$ is important to the following analysis. If $\lambda_i$ is the shadow value of the constraint that the violation is nonnegative for source $i$, this condition is

$$(2) \qquad p - [D'(v_i)(F(v_i) + p \times v_i) + D(v_i)(F'(v_i) + p)] + \lambda_i = 0.$$

If $e_i - q_i$ is strictly positive (i.e., the emission source does not fully comply), then $\lambda_i = 0$. The term in brackets is the marginal expected penalty. Thus, equation (2) implies that a partially compliant emitter sets its marginal expected penalty equal to the price. For an emitter to consider full compliance, the price must be less than the marginal expected penalty (because $\lambda_i > 0$).

To simplify this equation for the following applications, assume that the probability of detection is constant and equal to $d$ (so $D'(v_i) = 0$). In addition, assume the fine is just a fixed amount per unit of violation, so $F'(v_i) = f$. This sort of fine is used in several emissions trading systems (see table 13.1). Thus, the first-order condition (2) becomes:

$$(3) \qquad p + \lambda_i = d \times (f + p).$$

### 13.1.2 The Government's Choices

The government influences the private compliance decision through both the probability of detection, $d$, and the cost of a violation, $f$. The probability of detection, $d$, depends on the level and distribution of public monitoring resources. However, nongovernmental actors may also affect $d$. Whistleblowers, often employees of noncompliant firms, account for a high share of substantive environmental violations detected (Heyes and Kapur 2009). In addition, nonprofit environmental organizations play a substantial role in detecting violations of current environmental laws (Thompson 2000).[2]

2. Both of these forms of private enforcement are likely to result in a probability of detection that rises with the violation and, thus, a higher marginal expected penalty than assumed in the simplified condition in equation (3).

The government also has some control over the penalty, *f*. As Becker (1968) famously argued, high fines can substitute for costly monitoring in raising the expected penalty. However, high fines are rarely used in practice. The reasons may include horizontal equity concerns and judgment-proof problems (firms cannot be fined more than the depth of their pockets). The government may face political obstacles to imposing Draconian fines. Finally, high fines may trigger costly litigation, as violators have incentives to spend more to fight them.

In an emission trading system, non-Draconian fines can play the role of a "safety valve," allowing polluters to avoid buying permits during price spikes and, thus, effectively setting a ceiling on the marginal cost of carbon reductions (Montero 2002; Kruger and Pizer 2004). However, a requirement that facilities forfeit missing allowances discourages the use of fines as a safety valve. To use fines as a safety valve, the government might eliminate this requirement or allow the emitter to delay forfeiting allowances until allowance prices fall.

A Beckerian high-fine regime could also produce a low expected marginal penalty that could act as a safety valve if the government chooses a low enough *d*. In such a regime, polluters would not disclose their violations and would face a small risk of high fines. Although it would lower the government's enforcement costs, such a regime would be less transparent than a fine set as an explicit safety valve.

### 13.1.3 Penalties and Compliance in Practice

Fines in emission trading programs have mostly been modest in practice. Table 13.1 presents a summary of fines in the EU ETS and the US $SO_2$ allowance program, with price information for scale.

Compliance with emission trading systems seems to have been high.[3] The United Kingdom reports no detected violations of the EU ETS from 2006 through 2008 and 99.7 percent compliance in 2005 (UK Department of Energy 2009). Landgrebe (2009) suggests the following numbers of German facilities with some sort of violation, relative to a total of 1,665 facilities issued allowances: 2005, 174 installations; 2006, 28 installations; 2007, 20 installations. Kruger and Egenhofer (2006) report only twenty-one excess emissions penalties under the US $SO_2$ allowance program in its first ten years.[4]

High compliance rates are something of a puzzle because of the low level

3. Two frauds recently perpetrated on the EU ETS are exceptions. One scam exploited cross-border collection of the EU value added tax (VAT); the perpetrators purchased allowances without paying the VAT and then resold them, claiming to collect tax they actually pocketed (Europol 2010). A "phishing" scam also targeted the EU ETS (Kanter 2010). However, neither fraud seems to reveal an enforcement problem specific to climate policy.

4. RECLAIM is an exception to the high compliance rates with 85 to 95 percent compliance in early years. Stranlund, Chavez, and Field (2002) attribute the lower compliance to penalties that are less automatic and to higher prices relative to penalties.

**Table 13.1** **Penalties with comparison to allowances prices**

| Program | Fine | Forfeit next period? | Allowance price Average | Maximum |
|---|---|---|---|---|
| EU ETS, 2005–2007 | €40 | Yes | €18 | €30 |
| EU ETS, 2008–2012 | €100 | Yes | €17 | €29 |
| US $SO_2$ allowance program (in 2008) | $3,337 | Yes | $380 | $550 |

*Notes:* European Union (EU) Emission Trading System (ETS) prices calculated from BlueNext are for 2006 (first trading period) and for 2008–2009 (second trading period). The $SO_2$ fine is adjusted for inflation, from a base of $2,000 in 1990 dollars. $SO_2$ prices in the table are approximate.

of fines. To assure complete compliance, the first-order condition (2) implies that the marginal expected penalty must exceed the price. With the simplifying assumptions behind equation (3), full compliance requires $p < d \times (f + p)$. For the first trading period of the EU ETS, the penalty for a violation was €40. Therefore, if we believe compliance was, in fact, virtually complete, detection rates had to be greater than $d = 18/(40 + 18)$, or 31 percent, at the average price of €18. At the peak price of €30, they had to exceed 43 percent. The necessary probabilities would have declined with the higher penalties in the second period, but would still have been high.[5]

The perceived chance of detection seems unlikely to be so high, particularly for small violations.[6] Perhaps widespread violations do occur but are not detected. More likely, firms expect costs from noncompliance in addition to the official fines, so the preceding calculations understate the private costs of noncompliance. Noncompliance may tarnish the firm's image with its consumers, host community, potential employees, and regulators. These concerns may loom especially large in a carbon market with ongoing government allocation of valuable allowances: the participants may worry that current noncompliance will lower their future allowance allocations.

If firms perceive a large informal penalty, full compliance requires a lower risk of detection, $d$, than it would have required with official fines only. The possibility of substantial informal penalties has two policy implications. First, if the government faces constraints on the magnitude of official fines, it might try to raise informal penalties. For example, press releases with the names of violators might draw attention, lowering the required $d$ and, thus, the government's enforcement costs. Second, high informal penalties make it difficult for the government to use fines as a safety valve. Even if the official

5. Stranlund, Chavez, and Field (2002) conduct similar calculations of required detection rates for the $SO_2$ allowance program.

6. Perceived chances of detection may dramatically overstate the reality. Research on income tax compliance shows households consistently overestimate their risk of an audit (Andreoni, Erard, and Feinstein 1998). However, the large firms involved in carbon emissions are likely to be more savvy about actual monitoring systems and detection risks.

fines are low enough to provide a safety valve at a relevant price level, firms may still have strong incentives to comply because of these other costs of violation.

## 13.2 Heterogeneous Monitoring Costs

Relative to the enforcement problems that have been studied previously, carbon markets add the complication of especially heterogeneous monitoring costs. Because such heterogeneous costs may raise novel issues for policy design, this section presents information on the cost differential for market participants. Information on costs for public enforcement agencies is not available but likely shows the same sort of variation as private monitoring costs.

### 13.2.1 Direct Costs of Monitoring

Large facilities that emit carbon dioxide probably do not face high costs when trying to demonstrate compliance. As with air pollution generally, the government may allow facilities to demonstrate compliance either through mass-balance approaches or continuous emissions monitoring (CEM). With a mass-balance approach, the facility uses the characteristics of its inputs and production technology to infer pollution without directly monitoring it. For carbon dioxide emissions from most large sources, this inexpensive approach yields an accurate accounting of emissions. The EU ETS allows many types of sources to use this approach to establish compliance (European Commission 2007).

The alternative, CEM, involves equipment that measures facilities' releases directly. The US Clean Air Act already requires CEM of $CO_2$ for the large coal-fired power plants that account for vast majority of $CO_2$ from power plants (Ackerman and Sundquist 2008). Thus, even the choice to rely extensively on CEM for enforcement of a US climate policy would probably not generate large new costs.[7]

The EU ETS requires third-party verification of emissions from facilities subject to its controls. This approach partially privatizes enforcement and creates a system analogous to the verification system for offsets. A verification market participant reports that "verification costs ranged from €5,000–€7,500 . . . for a simple site to €10,000–€20,000 . . . or more for a more complex site" (Kruger and Pizer 2004, 19) in the voluntary UK Emissions Trading Scheme, which ran from 2002 to 2006. Third-party verifica-

7. The US $SO_2$ allowance program requires CEM for large sources, although facilities could probably have calculated emissions with fairly high precision. Ellerman et al. (2000) find that CEM has been costly, contributing to private monitoring costs equal to 7 percent of total compliance costs. However, they argue that this approach has the advantage of separating true compliance activities from monitoring and helped convince skeptics of the environmental effectiveness of tradable permit programs.

tion probably raises social costs by less than this amount, however, because verification substitutes for public monitoring and for activities the source might have conducted internally.

A survey by Jaraite, Convery, and Di Maria (2009) of Irish firms in the EU ETS first trading period provides data on overall private monitoring costs. It finds that "monitoring, reporting, and verification" (MRV) costs averaged €0.04 per ton of $CO_2$ or about €25,000 per year per respondent. Thus, monitoring costs averaged only about 0.1 percent of the total compliance costs, if we assume average compliance costs are a quarter of marginal costs (the allowance price).[8] Jaraite, Convery, and Di Maria also report that 40 percent of MRV costs are for external consultants, which confirms the market participant report from Kruger and Pizer (2004).

Private monitoring and verification costs for other sources, such as those proposed as the basis for offsets, are probably much higher for several reasons. The emissions may not be from point sources, raising challenges for any direct measurement of releases. Verifying all the values necessary to calculate greenhouse gas reductions from nonpoint sources may be complex. For example, the net effect of land use changes on greenhouse gas concentrations may vary greatly with specific agricultural or forestry practices and characteristics of the land. Originators of offsets may bear the burden of establishing "additionality," that is, that pollution is reduced relative to some meaningful baseline (Montero 1999; Bushnell, chapter 12 in this volume; Hahn and Richards, Forthcoming). Finally, the relevant activities may take place abroad and possibly in countries with more corruption, adding to the complexity of assuring compliance.

Antinori and Sathaye (2007) provide an estimate of monitoring and verification costs for offsets. The twenty-eight greenhouse gas reduction projects that they study report average monitoring and verifications costs of $0.30 per metric ton of $CO_2$, although the variance is high and a few large projects report much lower costs.[9] When compared to the average monitoring costs of €0.04 (about $0.06) per ton of $CO_2$ for covered facilities in Jaraite, Convery, and Di Maria (2009), these estimates suggest monitoring costs may be several times higher for offsets than for covered facilities.

8. Ellerman, Convery, and De Perthuis (2010) report a lack of ex post estimates of the total costs of the EU ETS first trading period and assume the average costs are half the marginal costs (a linear marginal cost curve).

9. Alternatively, one can estimate the costs of monitoring offsets by comparing prices for emissions reductions with more rigorous and less rigorous certification. Conte and Kotchen (2010) analyze carbon offset prices from an online listing in 2007, 13 percent of which have the more rigorous certification that would make them eligible for use in compliance with Kyoto obligations and the EU ETS. They estimate that certified permits cost 30 percent more than other projects with similar observable characteristics. Although many demand and supply factors may underlie this price differential, the costs of the certification probably contribute part of it. If even 10 percent of Conte and Kotchen's low-end estimate of a 30 percent price difference is monitoring costs, these costs are $0.54 per ton of $CO_2$.

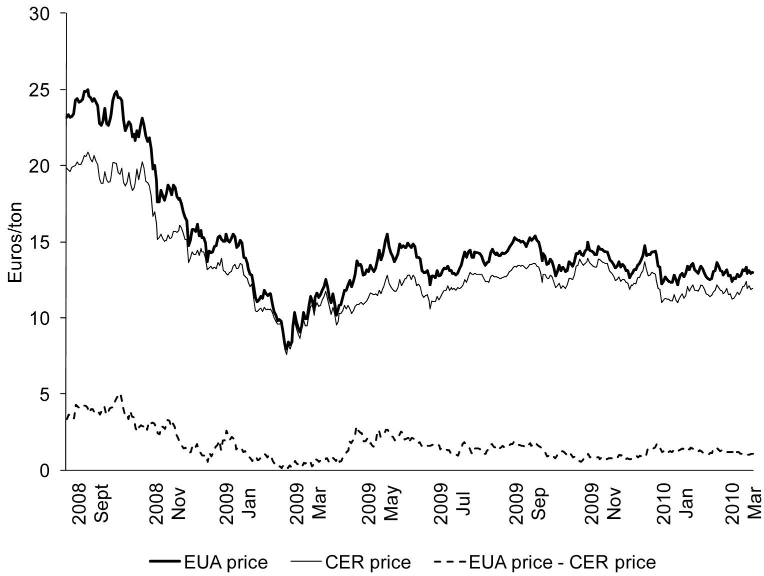


**Fig. 13.1 Spot prices of European Union Allowances (EUAs) and secondary Certified Emissions Reductions (CERs) and their difference**

*Source:* BlueNext (www.bluenext.fr).

### 13.2.2 Differential Enforcement Risks

Higher monitoring costs probably reduce private monitoring. With less thorough monitoring, allowances may be subject to greater risk that the government will find them invalid and conclude that the emitter is out of compliance. The variation in private monitoring costs may lead to variation in what I will call the "validity" of the allowance: the chance that the emitter is deemed to be in compliance when using that allowance. Market prices may reflect any differences in validity across different sorts of allowances and, thus, provide indirect evidence of differential monitoring costs.

Figure 13.1 presents the history of the differential between two types of allowances in Europe. Facilities subject to EU ETS restrictions may cover their emissions either with the European Union Allowances (EUAs), which the EU issues to point sources of $CO_2$, or with Certified Emissions Reductions (CERs). CERs result from greenhouse gas emission reduction projects undertaken through the Kyoto Protocol's Joint Implementation (JI) or Clean Development Mechanism (CDM).[10]

The figure compares spot market prices of EUAs and "secondary" CERs

10. The vast majority of CERs originate in China and derive from hydroelectric and wind projects (Capoor and Ambrosi 2009).

on one of the major exchanges, BlueNext.[11] "Secondary" CERs are being resold, as opposed to "primary" CERs sold by the originating project. The average price differential between EUAs and CERs from August 2008 through February 2010 is €1.64; the maximum of €5.03 occurred early in the period when allowance prices were highest.

We would expect EUAs and CERs to be perfect substitutes for complying EU facilities; thus, the existence of a price difference requires explanation.[12] One possibility is that the public relations consequences of using EUA and CERs differ, even if the two types of allowances are equally valid from an enforcement perspective. The public may view CERs less favorably than EUAs because CERs relax the constraint that Europe has put upon its own carbon dioxide emissions. However, the public also might prefer CERs to EUAs as "charismatic carbon"; CERs may promise nonclimate benefits, such as reducing local air pollution or protecting natural ecosystems. A second possibility is that market participants perceive a greater risk of being found out of compliance with CERs than with EUAs. The price differential then measures the disparity in the expected validity of the two types of allowances.

Suppose the risk that the government finds a violation is $d_{\mathrm{EUA}}$ for EUAs and $d_{\mathrm{CER}}$ for CERs. The penalty is the same with either type of permit because it consists of a fine and forfeit of EUAs from next year's allocation. Using the simplified first-order condition in equation (3), the difference in the marginal expected penalties and, thus, the price premium is $p_{\mathrm{EUA}} - p_{\mathrm{CER}} = (d_{\mathrm{EUA}} - d_{\mathrm{CER}}) \times (f + p_{\mathrm{EUA}})$. With the official fine of $f =$ €100, an average EUA price of €25 over the period of price premium data, and an average premium of €1.64, the detection probabilities would differ by 1.3 percentage points, a modest amount.

However, a major objection to this calculation is that the EU ETS places liability for compliance on sellers. Thus, the buyer of CERs might not believe it faced any higher expected penalty than if it had purchased EUAs. On the other hand, public opinion may not respect the legal allocation of compliance obligations, so a violation may still have public relations costs for the buyer. Depending on the comparison between the marginal public-relations cost and the official fine, the 1.3 percentage point disparity may be either too high or too low.

### 13.2.3 Policy Design with Heterogeneous Enforcement Costs

The variation in monitoring costs across different sources of allowances (e.g., the EUA-CER differential) and the resulting differences in validity give

11. Mizrach (2010) discusses the exchanges and analyzes various spot and futures prices in international carbon markets, including the EUA-CER spread.

12. The EU ETS does place caps on the number of CERs each country may use cumulatively over the second trading period. However, this country-level constraint does not affect an individual source's current ability to substitute freely between the two types of allowance and, thus, does not imply different current spot prices.

rise to a number of questions about policy design. One question is whether sources of allowances with high monitoring and enforcement costs ought to be excluded from the market. For example, a US climate policy might allow only domestic offsets or no offsets at all.

The simple enforcement model suggests that broadening the program might not reduce the compliance rate, despite spreading enforcement resources more thinly. Expanding the possible sources of allowances brings additional low cost sources of greenhouse gas abatement into the market, lowering the price of allowances. This reduction in price means that the marginal expected penalty required for full compliance falls and, thus, a lower detection rate can sustain full compliance.[13] Sigman and Chang (2011) present an expression for the conditions under which allowing offsets increases compliance: the effect on compliance depends on the relative costs of abatement and of auditing compliance in the capped and offset sectors and the number of offsets claimed.[14]

To illustrate the possible magnitude of the effect, consider a broadening of the market that causes the allowance price to fall from $p$ to $\delta p$. Using equation (3), the probability of detection required for full compliance falls from $d_0 = p/(f + p)$ to $d_1 = \delta p/(f + \delta p)$. For example, the US Environmental Protection Agency (2009) estimates that elimination of international offsets would nearly double the allowance price (from \$13–\$17 to \$25–\$33 in 2015; from \$17–\$22 to \$33–\$44 in 2020) for the Waxman-Markey climate policy. If the fine were set at five times the initial allowance price (along the lines of the EU ETS), including the international allowances would allow the $d$ required for full compliance to fall to 55 percent of the $d$ in the narrower market.[15]

The net effect on compliance depends upon the relationship between government outlays and $d$ in the narrower and broader markets. Obviously, if detection is too difficult and fraud rampant in the broader market, overall compliance will decline with the expansion. Nonetheless, the reduction in the required detection rate does suggest at least the possibility that market expansion improves compliance, despite apparent enforcement difficulties.[16]

13. An analogous effect might arise with nonenvironmental taxes. Lower marginal tax rates decrease the incentive to evade taxes (Allingham and Sandmo 1972; Clotfelter 1983). Thus, a revenue-neutral tax reform that reduces marginal tax rates by broadening the tax base might improve compliance, even if it increased the difficulty of monitoring all taxed activities.

14. Sigman and Chang (2011) also point out that costly enforcement may be a reason to use offsets, rather than expanding the cap to include the second sector: the government need only audit claimed offsets, not the entire sector.

15. This example illustrates possible magnitudes only. The actual Waxman-Markey legislation set the excess emission fine at twice the allowance price (H.R. 2454, 111th Congress, Section 723). This rule would reduce compliance incentives along with compliance costs and not give rise to the effect in the text.

16. This analysis takes a narrow view of "compliance" for offsets, considering only whether actions promised are undertaken, not whether they contribute to an overall reduction in atmospheric greenhouse gases. Elsewhere in this volume, Bushnell (chapter 12) and Borenstein (chapter 6) consider broader issues in expanding the sources of greenhouse gas abatement.

### 13.3 Conclusions

A climate policy that controls domestic $CO_2$ emissions from fossil fuels may not present too great an enforcement challenge. Experiences with the EU ETS and the US $SO_2$ trading program suggest a high degree of compliance with emission trading, despite modest penalties. High compliance may partly result from public relations costs for violators.

Previous experience suggests that monitoring costs vary substantially across different types of allowances in current markets. This variation raises some interesting questions for future analysis. For example, it would be useful to study whether enforcement agencies could improve the overall efficiency of the program by narrowing the difference in the validity of allowances from different sources.

A policy response to the variation in enforcement costs could be to restrict the market to areas of low enforcement cost. However, the simple model presented here suggests that expanding the market to include activities that require more costly enforcement may not lower compliance if inclusion of these abatement opportunities reduces allowance prices sufficiently. This analysis shows the importance of recognizing that enforcement strategies can respond to market conditions and that market conditions may be sensitive to these strategies. Both aspects of this relationship deserve additional consideration in climate policy design.

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## Comment Severin Borenstein

Hilary Sigman does an excellent job of presenting both a compelling theoretical argument and some interesting data on the impact of enforcement and detection in tradeable pollution permit markets. The conclusion that extending the market to areas with lower detection rates could actually raise compliance rates is particularly thought-provoking. For me, it provoked thoughts about optimal combinations or separations of markets. In particular, while it might make sense to include uncovered polluters in an existing market even if it is more difficult to detect cheating among the new participants, I believe it can also make sense to establish separate markets for participants with differential detection probabilities.

Consider an exisiting emissions market in which the probability of detection, $d_1$, and the fine for failing to purchase sufficient permits, $f_1$, are such that there is perfect compliance among all emitters. For the purpose of this intuitive discussion, assume that enforcement costs are zero, and detection rates are purely exogenous. Assume that the equilibrium permit price in that market is $p$. Now consider a second set of emitters who, in aggregate, have exactly the same abatement cost curve as in the first market, but may have a different probability of being detected if they purchase fewer emission permits than their actual emissions, $d_2$, could differ from $d_1$. The fine for detection is the same in both markets, $f_2 = f_1 = f$. There are (at least) three possible treatments of this second set of emitters: (a) include them in the existing emissions market, (b) establish a separate emissions market for the second set of emitters, or (c) do not regulate the second set of emitters at all. With zero enforcement costs, option (b) clearly dominates option (c). The comparison of options (a) and (b) is more interesting, however.

Consider expanding the permit market to include the second market while

Severin Borenstein is the E. T. Grether Professor of Business Economics and Public Policy at the Haas School of Business, University of California, Berkeley; codirector of the Energy Institute at Haas; director of the University of California Energy Institute; and a research associate of the National Bureau of Economic Research.

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