PRELIMINARY
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Monitoring and Enforcement of Climate Policy

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ABSTRACT: This paper applies recent research on environmental enforcement to a potential U.S. program to control greenhouse gases, especially through emission trading. Climate change policies present the novel problem of integrating emissions reductions from sources that are relatively easy to monitor (such as carbon emissions from fossil fuels) with sources with sources that are very difficult to monitor (such as other greenhouse gas emissions). The paper documents the heterogeneity in monitoring costs across different parts of the carbon market. It argues, however, that a broad emission trading system that includes more difficult-to-enforce components can have higher overall compliance than a narrower program.

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Effective enforcement is critical to a successful climate change policy.[1] Fortunately, it should be possible to enforce many of the central elements of a climate change policy transparently and at a reasonable cost. However, enforcement of other aspects of climate policy can be daunting. Enforcement is a dominant consideration in the design of policies to address either greenhouse gases other than carbon dioxide or reductions in carbon dioxide from sources other than fossil fuels.

As a consequence, climate change policy poses the novel challenge of integrating easy-to-enforce and difficult-to-enforce components in one policy. In this chapter, I discuss this issue and present some data from existing carbon markets on the disparities in ease of enforcement. Empirically, enforcement costs seem to differ considerably across compliance methods. However, with emissions trading, expanding compliance to include more difficult to enforce policies may reduce the enforcement challenge by lowering allowance prices.

1 Incentives for compliance with a climate policy

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

1.1 The compliance decision

The standard environmental enforcement model considers a risk-neutral emitter who seeks to minimize its cost of complying with the regulation plus the expected punishment.[2]

Compliance costs depend on the form of the public policy. For a performance standard, these costs are just the cost of reducing emissions, $c(e_i, \gamma_i)$, where e_i is the emission choice by i and

[1] A few recent papers address the economics of the enforcement of climate change policies specifically (Kruger and Pizer, 2004; Johnstone, 2005; Kruger and Engenhofer, 2006).

[2] 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.

γ_i reflects cost heterogeneity. With a emissions trading or carbon tax, the compliance cost also includes a term that reflects outlays for purchases (or income from sales) of permits or tax paid on emissions. For a cap-and-trade program with initial allowance allocation of Q_i, the compliance cost is then the sum of pollution reduction costs and net purchases of permits, $c(e_i, \gamma_i) + p * (q_i - Q_i)$, where p is the equilibrium permit price and q_i the quantity of permits the emitter applies to its own emissions. The emitter thus has choices on two margins: its emissions, e_i and its supply or demand for permits, q_i. A carbon tax is little different: the polluter choses a level of pollution to report as its tax base, instead of a number of permits q_i, and gets no initial allocation.

The expected penalty consists of a chance of the violation is detected, d, and a fine, F, which may depend on the magnitude of the violation. For a emission trading system, the violation is difference between the emissions and the permits used, $e_i - q_i$. In addition to the fine, most policies require that the violator "fix" the violation. This requirement attempts to strengthen the assurance that violating the law is not 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: $F(e_i - q_i) + p * (e_i - q_i)$ (ignoring discounting if permits are surrendered next year). This 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 non-negative:

$$\begin{aligned} \min_{e_i, q_i} \quad & c(e_i, \gamma_i) + p * (q_i - Q_i) + d(e_i - q_i)[F(e_i - q_i) + p(e_i - q_i)] \\ \text{s. t.} \quad & e_i - q_i \geq 0 \end{aligned} \tag{1}$$

The first order condition with respect to q_i is important in the analysis below. If λ_i is the shadow value of the constraint that the violation is non-negative for source i, this condition is

$$p - [d'(v_i)(F(v_i) + p * v_i) + d(v_i)(F'(v_i) + p)] + \lambda_i = 0, \tag{2}$$

where $v_i = e_i - q_i$. The term in brackets is the marginal expected penalty. Equation (2) implies that an emitter sets its marginal expected penalty equal to the price, if it has chooses an interior solution for the violation (i.e., does not fully comply).

To simplify the equation for applications, assume that the probability of detection is constant ($d'(v_i) = 0$) and that 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 1). Thus, the condition becomes

$$p + \lambda_i = d * (f + p). \tag{3}$$

1.2 The government's choices

The government has some control over both d and f, but its control is incomplete. The probability of detection, d, does depend in part on the level and distribution of public monitoring resources. Nongovernmental forces may also be important to d. Whistle-blowers, often employees of non-compliant firms, account for a high share of substantive environmental violations detected (Heyes and Kapur, 2009). In addition, non-profit environmental organizations play a substantial role in detecting environmental problems (Thompson, 2000). Both of these forces may be even more influential in climate change enforcement because of the moral imperative that many people attach to greenhouse gas abatement.[3]

The government also has some control over the penalty, f. The government can assure complete compliance with a sufficiently high expected penalty. 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 judgment-proof problems (firms cannot be fined more than the depth of their pockets) and horizontal equity concerns. The government may face political obstacles in trying impose draconian fines. Finally, high fines may trigger costly litigation as violaters have incentives to spend more to fight high fines.

[3]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 equation (3).

Table 1: Penalties with comparison to allowances prices

Program	Fine	Forfeit next period?	Allowance price Average	Maximum
EU ETS, 2005-07	€40	Yes	€18	€30
EU ETS, 2008-12	€100	Yes	€17	€29
US SO_2 allowance program (in 2008)	$3,337	Yes	$ 380	$550

Notes: EU ETS prices calculated from BlueNext are for 2006 (first trading period) and for 2008-2009 (second trading period); SO_2 fine is adjusted for inflation, from a base of $2000 1990 dollars. SO_2 prices are approximate.

For an emissions trading program, 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 marginal cost ceiling on carbon reductions (Montero, 2002; Kruger and Pizer, 2004). For the safety-valve to be effective, facilities should not be required to forfeit missing allowances, unless the government allows them to wait until prices fall to do so.

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 face a small risk of high fines. A policy in which polluters would declare their use of the fine as a safety-valve seems preferable, however, from the perspective of transparency and respect for the law.

1.3 Penalties and compliance in practice

Fines in emission trading programs have mostly been modest. Table 1 presents a summary of fines in the EU Emissions Trading System (EU ETS) and in some other programs, with price information for scale.

Compliance with the EU ETS and other emission trading programs seems to have been high.[4] The UK reports no detected violations of the EU ETS from 2006 through 2008 and 99.7% com-

[4]Two frauds recently perpetrated on the EU ETS may be exceptions. One scam exploited cross-border collection of the EU VAT; the culprits purchased allowances without paying the VAT and then resold them, claiming to collect tax they actually pocketed (Europol, 2009). A "phishing" scam also targeted the EU ETS (Kanter, 2010). Despite some overheated press responses, neither fraud reveals a fundamental enforcement problem for climate policy itself.

pliance in 2005 (U.K. 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 21 excess emissions penalties under the US SO_2 Allowance program in its first ten years.[5]

High compliance rates are something of a puzzle because of the low level 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 $p < d * (f + p)$. For the first trading period of the EU ETS, the penalty for a violation was €40. Therefore, at the average price of €18, detection rates had to be greater than $\frac{18}{40+18}$ or 31% and, at the peak price of €30, they had to exceed 43%, if we believe compliance was in fact virtually complete. The necessary probabilities would have declined with the higher penalties in the second period, but would still have been high.[6]

The perceived chance of detection seems unlikely to be so high, particularly for small violations.[7] Perhaps widespread violations do occur, but are not detected. More likely, firms expect costs from noncompliance other than the official fines, so the calculations above understate the private costs of noncompliance. Noncompliance may tarnish the firm's image with its consumers, host communities, potential employees, and regulators. If firms perceive a large unofficial penalty, they will require a lower risk of detection, d, for full compliance.

One implication of the large official penalty that these figures imply is that the government will have difficulty using official fines as a safety valve. Even if the official fines are set very low, firms may still have strong incentives to comply because of these other costs.

[5]RECLAIM is an exception to the high compliance rates with 85-95% compliance in early years. Stranlund et al. (2002) attribute the lower compliance to penalties that are less automatic and to higher prices relative to penalties.

[6]Stranlund et al. (2002) conduct similar calculations of required detection rates for the SO_2 Allowance program.

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

2 Heterogeneous monitoring costs

Relative to the enforcement problems that have been studied in the previous literature, 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.

2.1 Direct costs of monitoring

Large facilities that emit carbon dioxide probably do not experience high costs to demonstrate compliance. They may calculate carbon emissions using mass-balance approaches or may use continuous emissions monitoring (CEM).[8] The EU ETS allows emissions to be calculated from inputs and production technology for many sources (EC, 2007). In the US, a number of firms already have installed CEM for CO_2. Forty percent of firms reporting CO_2 to the EPA use CEMs (Kruger and Engenhofer, 2006). Thus, even with more agressive emission monitoring requirements, the US probably will not experience especially high costs of monitoring CO_2 emissions from point sources.

The EU ETS probably raises these monitoring costs by requiring third-party verification of emissions. This approach partially privatizes the enforcement processes and creates a system that is more directly analogous to the verification system for offsets. Citing an verification market participant, Kruger and Pizer (2004) report 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" (p. 19) in the voluntary UK Emissions Trading Scheme, which ran from 2002 to 2006. Third-party verification probably raises social costs by less than this amount, however, because verification substitutes for activities the source might have conducted internally and for public monitoring.

[8]The US SO_2 Allowance program requires CEM for large sources, although releases could probably have been adequately calculated. Ellerman et al. (2000) find that CEM has been costly, contributing to private monitoring costs equal to 7% 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.

The best available data on the overall private monitoring costs are from a survey by Jaraite et al. (2009) of Irish firms in the EU ETS first trading period. They report that "monitoring, reporting and verification" (MRV) costs averaged €0.04 per ton of CO_2 or about €25,000 per year per respondent. As a very rough estimate, if the average abatement costs are a quarter of the price, then monitoring costs averaged only about 1% of costs. Jaraite et al. also report that 40% of MRV costs are for external consultants. The latter value corresponds fairly closely with the €10,000 per facility that Kruger and Pizer (2004) report for external verification.

Private monitoring and verification costs for sources outside the market, such as those proposed as offsets, are probably much higher for several reasons. Offsets may derive from many different activities. The emissions may not be from point sources, ruling out continuous emissions monitoring and requiring more complicated information. The burden of establishing "additionality" (reductions relative to some meaningful baseline) also may fall on originators of offsets. Finally, the relevant activities may take place abroad and possibly in countries with more corruption, adding to the complexity of assuring compliance.

Direct information on the monitoring costs for offsets is not available. However, an indication of the cost of verification for offsets may be found in the higher prices paid for Certified Emissions Reductions (CERs) relative to other carbon reductions in the voluntary carbon market. CERs result from projects undertaken through the Kyoto Protocol's Joint Implementation (JI) or Clean Development Mechanism (CDM) and thus require a high-standard of third-party verification and monitoring. The vast majority of CERs originate in China, with hydro and wind projects as the largest activities (Capoor and Ambrosi, 2009). Some demand for emission reductions also comes from sources that do not require JI/CDM certification. These include individuals and firms who voluntarily offset their carbon footprint. Thus, it is possible to compare the prices for emissions reductions with more rigorous and less rigorous certification.

Conte and Kotchen (2009) conduct an analysis of carbon offset prices in an on-line listing in 2007, 13% of which were JI/CDM certified. They estimate that the certified permits cost between 30 and 65% 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% of Conte and Kotchen's low-end estimate of a 30% price difference is monitoring costs, these costs are \$0.54 per ton of CO_2 for CERs. By comparison, the Irish study, Jaraite et al. (2009), suggests average monitoring costs of €.04 per ton of CO_2 for covered facilities. Thus, the costs may differ by an order of magnitude for these different allowance sources.

2.2 Differential enforcement risks

The higher the private monitoring costs are, the less thorough private monitoring is likely to be. 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 shown above thus may lead to variation in what I will call the "validity" of the allowance: the chance that the emitter will be deemed to have complied if it uses that allowance. Market prices should reflect any differences in validity across different sorts of allowances and, thus, provide indirect evidence of differential monitoring costs.

Figure 1 presents the history of the premium between two types of allowances in Europe. Facilities subject to EU ETS restrictions may use cover their emissions either using European Union Allowances (EUAs), the allowances issued to carbon emitters, or using CERs. The figure compares spot market prices of EUAs and CERs on one of the major exchanges, BlueNext.[9] The CER price is for "secondary" CERs, emissions reductions not sold by the originating project. The average price differential from August 2008 through February 2010 is €1.64; the maximum of €5.03 occurs early in the period when allowance prices were highest.

EUAs and CERs should be perfect substitutes for complying EU facilities; thus the existence of a price difference requires explanation.[10] One possibility is that the public relations consequences

[9] Mizrach (2009) discusses the exchanges and analyzes various spot and futures prices in international carbon markets, including the EUA-CER spread.

[10] 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 individual sources' current ability to substitute freely between the two types of allowance at present and thus does not imply different current spot prices.

Source: BlueNext (www.bluenext.fr)

Figure 1: Spot prices of EUAs and CERs and their difference

of using EUA and CERs differ, even if the two types of allowances are equally valid from an enforcement perspective. CERs may be more suspect than EUAs because they 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 non-climate benefits, such as reducing local air pollution.

A second possibility is that market participants perceive a greater risk of being found out of compliance with CERs than EUAs. The price differential measures a disparity in expected validity of the two types of allowances, if prices are determined by firms that do not fully comply.

Suppose the risk that the governments finds a violation is d_{EUA} for EUAs and d_{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 $p_{EUA} - p_{CER} = (d_{EUA} - d_{CER}) * (f + p_{EUA})$. 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 enforcement probabilities should 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 turn on the legal allocation of compliance obligations, so a violation may still damage public relations. 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.

2.3 Policy design with heterogeneous enforcement costs

The variation in costs across different sources of allowances (e.g., the EUA-CER differential) and the resulting differences in validity produce a number of issues for policy design. A question for US climate policy is whether evidence of very high monitoring and enforcement costs for some sources of allowances is a reason to exclude them from the market. For example, a policy might allow only domestic offsets or no offsets at all.

The simple enforcement model above suggests, however, that even with a fixed enforcement budget, broadening the program might not reduce the compliance rate. Compliance could increase, even as fixed enforcement resources are spread more broadly, because the allowance price determines incentives for noncompliance. Expanding the possible sources of allowances brings additional low cost sources of greenhouse gas emissions into the market, lowering the price of allowances. This reduction in price means that the marginal expected penalty required for full compliance falls, allowing less monitoring effort.

Consider a broadening of the market that causes prices to fall from p to δp. Using the simplifying assumptions, the probability of detection, d, required to assure complete compliance falls from $\frac{p}{f+p}$ to $\frac{\delta p}{f+\delta p}$. For example, US EPA (2009) estimates that elimination of international offsets would nearly double the allowance price (from \$13–\$17 to \$25–\$33 in 2015 and from \$17–\$22 to \$33–\$44 in 2020) for the Waxman-Markey bill. If we assume a 5:1 ratio of fine to allowance price (along the lines the EU ETS), including the international allowances would allow the d required for full compliance to fall to 55% of the d required without the international allowances. Whether this change is sufficient to increase compliance would depend on the costs of enforcement in the narrower and broader markets and the shape of the relationship between enforcement spending and compliance. Nonetheless, it does suggest that it would be a mistake to rule out broader markets on enforcement grounds without further scrutiny.

3 Conclusions

Enforcement of a domestic climate change policy is probably not too great a challenge. Experience with the EU ETS and the trading programs in the US suggests a high degree of compliance, despite non-draconian penalties.

Previous experience does suggest that monitoring and enforcement costs for both private firms and public enforcement agencies vary substantially across different types of permits. This variation raises some interesting questions for future analysis. For example, when enforcement has significant costs, should enforcement agencies work to narrow the differences in the probabilities that allowances from different sources represent real emission reductions?

A policy response to the variation in enforcement costs would be to restrict the market to areas of low enforcement cost. However, the simple model presented here suggests that broader markets may not lower compliance if they allow lower allowance prices. This analysis shows the importance of recognizing that enforcement strategies should respond to market conditions and that market conditions will be sensitive to these strategies. Both directions of this relationship deserve additional study.

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