Abatement-Cost Heterogeneity and Anticipated Savings from Market-Based Environmental Policies

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Richard G. Newell and Robert N. Stavins*

1. Introduction

Over the past decade, policy makers in many parts of the world have given increasing attention to market-based instruments for environmental protection, including various types of tradeable permit and charge systems (Stavins 1999). Whereas market-based environmental policy instruments were controversial just ten years ago, they have now evolved in political circles to the point of becoming a conventional wisdom, at least in the United States (Keohane, Revesz, and Stavins 1998). This change may please many economists concerned with environmental policy, but it also highlights the importance of identifying the appropriate policy instrument for each environmental problem that is faced in its particular socio-economic context. In some cases, market-based instruments may be highly desirable, but in other cases their advantages may be relatively small. Our fundamental purpose is to provide some relatively simple rules-of-thumb for policy analysts and policy makers engaged in the early stages of exploring alternative policy instruments, with the hope that such rules-of-thumb can help identify policy instruments that merit more detailed investigation.

A variety of criteria can and have been brought to bear upon the choice of policy instruments to achieve environmental goals. As the stringency of environmental targets has

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increased, cost-effectiveness has become a more important criterion for instrument choice.[1] A key factor affecting relative aggregate abatement costs under alternative policy instruments is the heterogeneity of pollution control costs across sources. There are many reasons why the costs of complying with environmental regulations tend to be heterogeneous, including differences in plant location, size, age, and production technology. Location, for example, can affect costs due to differences in the quality and price of inputs (e.g., proximity to clean inputs), physical characteristics (e.g., urban or rural), and political jurisdiction (e.g., pre-existing regulations). While it is widely recognized that abatement-cost heterogeneity is a fundamental determinant of the potential cost-savings associated with market-based policy instruments, there is surprisingly little analysis of the relationship between the nature and magnitude of such heterogeneity and potential cost savings associated with these innovative policy instruments.

Although there is an absence of such studies, there is a relevant albeit small theoretical literature on the relationship between potential gains from trade and the underlying heterogeneity of consumer preferences and production technology. Most prominent in this literature are studies by: Weitzman (1977) and Suen (1990) on the effects of diversity in consumer preferences on the relative efficiency of the price system; Krueger and Sonnenschein (1967) on the relationship between price divergence across countries and the gains from international trade; and Mendlesohn (1986) on regulating pollution in the presence of heterogeneous benefits and costs. None provide a simple framework for directly estimating the cost-savings associated with using market-based policy instruments.

[1]For a list of additional candidate criteria for policy instrument choice, see Bohm and Russell (1985).

There is also a substantial empirical literature that explores the costs of using alternative environmental policy instruments for particular environmental problems.[2] These studies are concerned with specific environmental pollutants in specific contexts, ranging from dissolved oxygen in the Fox River of Wisconsin to particulates in Santiago, Chile. Each study uses different types of data at different levels of aggregation, each makes different simplifying assumptions, and each employs different methods of analysis. Although this differentiation in data and methods may be appropriate for the situations being studied, little intuition thereby emerges regarding the general relationship between the nature and magnitude of cost heterogeneity and the potential cost savings associated with cost-effective, market-based instruments.

The data requirements and analytical methods of many of the approaches utilized to compare the expected costs of alternative policy instruments render them inappropriate for the early stages of policy development. Instead, relatively simple rules-of-thumb that can be employed with minimal amounts of data may be preferable for conducting initial screenings of environmental problems, so that analysts and decision makers can focus their attention on cases where potential cost savings are greatest. Such screening is important because market-based instruments are by no means a panacea; in some cases, they hold tremendous promise of providing environmental protection cost effectively, but they are not well suited in other cases for a variety of reasons,[3] and there are significant political barriers to their adoption (Keohane, Revesz, and Stavins 1998). Therefore, it is important for policy analysts and policy makers to have some idea in

[2]See, for example: Atkinson 1983; Atkinson and Lewis 1974; Atkinson and Tietenberg 1982, 1991; Carlson, Burtraw, Cropper, and Palmer 1997; Coggins and Swinton 1996; Gollop and Roberts 1983, 1985; Hahn and Noll 1982; Kolstad 1986; Krupnick 1986; Maloney and Yandle 1984; McConnell and Schwartz 1992; O'Neil, David, Moore, and Joeres 1983; O'Ryan 1996; Perl and Dunbar 1982; and Seskin, Anderson, and Reid 1983.

[3]See, for example, Hahn 1984, Misolek and Elder 1989, Malueg 1990, and Stavins 1995.

advance of the potential cost savings that may be associated with using a market-based instrument for a particular environmental problem.

Our approach is to develop three highly stylized models of alternative means of achieving an aggregate environmental target: two command-and-control policy instruments and one market-based instrument. A fundamental feature of the models is that a subset of their parameters represent the nature and degree of abatement cost heterogeneity. We solve each model for the aggregate abatement cost it would imply, and then by comparing results, we derive expressions for the absolute and percentage cost-savings attributed to adopting a market-based instrument — our "rules-of-thumb." Subsequently, we demonstrate the potential use of these rules-of-thumb with a specific application using readily available information.

Any empirical cost-savings potentials that are identified with the approach we offer might be thought of as lower-bound estimates, because we abstract from several other dimensions along which the costs of market-based instruments are anticipated to be less than or equal to those of command-and-control approaches. First, firms can — in theory — reduce emissions through three types of activities: product output reduction; input substitution; and end-of-pipe abatement. We abstract from the first type of activity by treating output as exogenous, which seems reasonable, since pollution-control costs are typically a very small fraction of total production costs (Jaffe, Peterson, Portney, and Stavins 1995).[4] Further, we do not differentiate between input substitution and end-of-pipe treatments, considering both as emission abatement. Hence, our stylized command-and-control policy instruments are (uniform) performance standards, not technology mandates. This is important to recognize, since even if firms were

[4]There are, of course, exceptions to this, one prominent example being carbon dioxide emissions associated with electricity production.

perfectly homogeneous, a true technology mandate would not be cost-effective, since it provides no latitude for firms to substitute "cleaner" inputs.

Second, our focus is exclusively on static cost-effectiveness, but it is well known that the incentive-structure of market-based instruments can lead to "dynamic cost-effectiveness," that is, decreased abatement costs over time as a result of technological innovation and diffusion (Milliman and Prince 1989; Jaffe and Stavins 1995; Jung, Krutilla, and Boyd 1996; Newell, Jaffe, and Stavins 1999). Third and finally, our analysis is partial-equilibrium in nature. In a general-equilibrium context, particular types of market-based instruments, such as those that combine revenue generation with cuts in pre-existing distortionary taxes, can enjoy additional cost advantages over commensurate command-and-control regulations (Goulder, Parry, Williams, and Burtraw 1999).

In Section 2, we develop our stylized models of three basic types of policy instruments, carry out some comparative statics analysis, and identify key general findings. In Section 3, we apply the approach to the policy problem of reducing nitrogen oxide emissions in the northeastern United States, and in Section 4, we conclude.

2. A Model of Cost Heterogeneity and Policy Choice

2.1 A Model of Heterogeneous Abatement Costs

We posit the common situation in which the government seeks to limit the aggregate emissions of a set of sources. The output, χ_i, of each source is given, as is aggregate output, X, for all n sources. Emission quantities are given by q_i and are chosen by sources subject to policy constraints. Each source has a factor demand for pollution that is linear in the price, p, of pollution (relative to other factors) and proportional to output:

$$q_i = (\alpha_i - \beta_i p)\chi_i \,, \tag{1}$$

where α_i and β_i are source-specific demand parameters that allow for heterogeneity across sources in the intercept and slope of demand (and thus marginal cost).[5] For the sake of clarity, we assume the heterogeneous variables are independently distributed, with mean values *a*, *b*, and $\overline{x}$.[6]

The parameter α_i represents pollution intensity per unit of output when the price is zero, that is, the baseline level of pollution the source would choose in the absence of government regulation. The parameter β_i represents the rate at which the source alters its pollution intensity per unit change in the price of pollution. Solving for *p*, we can see that this demand function implies a marginal abatement cost function that is linear in abatement per unit of output:

$$p = -C'(q_i;\alpha_i,\beta_i,\chi_i) = \frac{1}{\beta_i}\left(\alpha_i - \frac{q_i}{\chi_i}\right) \tag{2}$$

where $C(q_i;\alpha_i,\beta_i,\chi_i)$ is the emissions cost function for each source. Thus, β is inversely proportional to the marginal cost of abatement, and different values of β alter the slope of the marginal abatement cost function. Integrating the negative of Equation (2) with respect to emissions, and assuming baseline emission costs are zero for each source, this structure implies an underlying cost function that is quadratic in emissions:

$$C(q_i;\alpha_i,\beta_i,\chi_i) = \frac{1}{2\beta_i}\left(\alpha_i^2\chi_i - 2\alpha_i q_i + \frac{q_i^2}{\chi_i}\right). \tag{3}$$

Henceforth, we refer to the emission cost function for each source simply as $C(q_i)$.

[5]See Table I for examples of units of the variables and parameters. In the simplest market model, one would anticipate that firms with relatively high abatement costs would be driven out of the market by firms with lower abatement costs. In reality, of course, this does not happen because those same firms may enjoy cost advantages along other dimensions, and because there are a variety of frictions in the relevant markets, including those due to the regulatory environment.

2.2 Alternative Policies for Emission Allocation

We consider three stylized policies for allocating aggregate emissions of Q among the sources. Two are performance-based instruments: a uniform emission rate standard and a uniform percentage reduction standard.[7] The third is a market-based instrument, such as an emissions fee or tradable permit system. The market-based instrument, if implemented under ideal conditions, will lead to a cost-effective allocation of the control burden among sources.

2.2.1 Cost of a Uniform Emission Rate Standard

A uniform emission rate standard results in emissions from each source of $\tilde{q}_i$ equal to aggregate emissions per unit of output, weighted by source output:

$$\tilde{q}_i = \left(\frac{Q}{X}\right)\chi_i \,. \tag{4}$$

where X is aggregate output. This assumes perfect compliance, an assumption we also make for the other stylized instruments we consider. We also assume that the standard is binding for the relevant set of sources; that is, each source undertakes non-negative reductions.

Substituting (4) into (3) we find the cost to each source of the uniform emission rate standard:

$$C(\tilde{q}_i) = \frac{1}{2\beta_i}\left(\alpha_i^2\chi_i - 2\alpha_i\left(\frac{Q}{X}\chi_i\right) + \frac{1}{\chi_i}\left(\frac{Q}{X}\chi_i\right)^2\right), \tag{5}$$

which upon taking expectations yields:

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[6]Allowing for correlation among the demand parameters would not change the preference *ordering* of market-based compared with command-and-control instruments, but it could alter the *magnitude* of the cost difference between these types of policies depending on the magnitude and sign of the correlations between α, β, and χ.

[7]We model the uniform performance standards as being in terms of an allowable emission rate per unit of product output, and a percentage reduction standard, because these are typical of command-and-control regulation. The model and the analysis is simpler for a uniform performance standard expressed in terms of abatement quantities, but less interesting in practical terms. The analysis is available from the authors upon request.

$$\mathrm{E}\left[C(\tilde{q}_i)\right]=\frac{1}{2}\mathrm{E}[\chi]\mathrm{E}[1/\beta]\left(\mathrm{E}\left[\alpha^2\right]-2\mathrm{E}[\alpha]\frac{Q}{X}+\left(\frac{Q}{X}\right)^2\right). \tag{6}$$

Recalling that $\mathrm{E}[\alpha]=a$ and $\mathrm{E}[\chi]=\bar{x}$, noting that $\mathrm{E}\left[\alpha^2\right]=a^2+\mathrm{V}[\alpha]$, and taking a second-order approximation of $\mathrm{E}[1/\beta]$ around b (the mean of β), we find

$$\mathrm{E}\left[C(\tilde{q}_i)\right]=\frac{\bar{x}}{2}\left(\frac{1}{b}+\frac{\mathrm{V}[\beta]}{b^3}\right)\left(\left(a-\frac{Q}{X}\right)^2+\mathrm{V}[\alpha]\right), \tag{7}$$

where $\mathrm{V}[\beta]$ is the variance of the slope of emissions demand.[8]

Multiplying by n, we find the aggregate expected cost of the uniform emission rate standard, denoted as $C\left(\tilde{Q}\right)$:

$$C\left[\tilde{Q}\right]=\frac{X}{2b}\left(1+\frac{\mathrm{V}[\beta]}{b^2}\right)\left(\left(a-\frac{Q}{X}\right)^2+\mathrm{V}[\alpha]\right). \tag{8}$$

Multiplying by a^2/a^2 we can rewrite (8) in terms of percentage reductions in emission rates:

$$C\left[\tilde{Q}\right]=\frac{Xa^2}{2b}\left(1+\frac{\mathrm{V}[\beta]}{b^2}\right)\left(R^2+\frac{\mathrm{V}[\alpha]}{a^2}\right). \tag{9}$$

where $R=\left(a-Q/X\right)/a$ is the percent aggregate reduction in emissions from the baseline to the aggregate emission constraint Q. We can further simplify this expression by recognizing that $\mathrm{V}[\beta]/b^2=v$ and $\mathrm{V}[\alpha]/a^2=\mu$ are dimensionless measures of heterogeneity or spread in α and β relative to their means, known as *coefficients of variation*. Upon substitution this yields:

$$C\left[\tilde{Q}\right]=\frac{Xa^2}{2b}(1+v)\left(R^2+\mu\right). \tag{10}$$

[8]Higher-order approximations of E[1/β] would involve terms for skewness, kurtosis, and higher-order moments of the distribution of β. We note that the skewness term would have a negative sign, indicating that a distribution with positive skewness (i.e., skewed to the right, or a long upper tail) would tend to have lower costs of a uniform

2.2.2 Cost of a Uniform Percentage Reduction Standard

A uniform percentage reduction standard results in emissions from each source of $\hat{q}_i$, where

$$\hat{q}_i = (1-R)\alpha_i\chi_i = \frac{1}{a}\left(\frac{Q}{X}\right)\alpha_i\chi_i . \tag{11}$$

Substituting (11) into (3) and taking expectations, we find the expected cost to each source of the uniform percentage reduction standard:

$$\mathrm{E}\left[C(\hat{q}_i)\right] = \frac{\bar{x}}{2b}\left(1+\frac{\mathrm{V}[\beta]}{b^2}\right)\left(1+\frac{\mathrm{V}[\alpha]}{a^2}\right)\left(a-\frac{Q}{X}\right)^2 . \tag{12}$$

Multiplying by n and writing the expression in terms of ν, μ, and R, we find the aggregate expected cost of the uniform percentage reduction standard, denoted as $C\left(\hat{Q}\right)$:

$$C\left[\hat{Q}\right] = \frac{Xa^2}{2b}(1+\nu)(1+\mu)R^2 . \tag{13}$$

2.2.3 Cost of a Market-Based Policy Instrument

Under a market-based instrument, a cost-effective allocation of emissions arises wherein each source chooses an emissions level q_i^* that minimizes its own abatement costs, so that

$$q_i^* = \left(\alpha_i - \beta_i p^*\right)\chi_i , \tag{14}$$

where an emissions market of size Q clears at price p^*. In principle, the cost-effective allocation could be implemented either by establishing a tradable permit market of size Q or by

performance standard. This makes intuitive sense since high values for β represent firms with low costs of emission control.

setting an emissions fee of p^*, where p^* will equal each source's marginal cost of emissions control in equilibrium.[9] Rewriting (14) in terms of rates of abatement,

$$\alpha_i - \frac{q_i^*}{\chi_i} = \beta_i p^*, \tag{15}$$

we note in passing that each source's cost-effective rate of abatement will be directly proportional to the slope of its emission demand function (or inversely proportional to the slope of its marginal abatement cost function). This implies that a source with twice as steep a marginal abatement cost function will undertake half as much rate reduction in a cost-effective equilibrium.

Taking expectations of (14), we find that average emissions will be demanded from a source with average levels of each characteristic:

$$\mathrm{E}\left[q^*(p^*)\right] = \bar{q} = \left(a - bp^*\right)\bar{x}, \tag{16}$$

where $\bar{q}$ represents average emissions. Solving for p^*, and substituting $Q/X = \bar{q}/\bar{x}$, we find the emissions tax that will deliver aggregate emissions of Q, which also equals the market-clearing permit price for a permit market of size Q:

$$p^* = \frac{a - Q/X}{b}. \tag{17}$$

Substituting Equation (17) back into Equation (14), we find that each source's cost-effective emissions level is given by:

$$q_i^* = \left(\alpha_i - \beta_i\left(\frac{a - Q/X}{b}\right)\right)\chi_i. \tag{18}$$

Following the same approach as before, we substitute (18) into (3) to find the cost to each source of the market-based instrument. Many terms cancel, yielding

[9] If Q is chosen optimally, p^* will also equal marginal benefits (avoided damages) at this level of emissions.

$$C\left(q_i^*\right)=\frac{\chi_i\beta_i}{2}\left(\frac{a-Q/X}{b}\right)^2. \tag{19}$$

Taking expectations and multiplying by *n*, we find the aggregate expected cost of the market-based instrument, denoted as $C\left(Q^*\right)$:

$$C\left(Q^*\right)=\frac{X}{2b}\left(a-\frac{Q}{X}\right)^2=\frac{Xa^2}{2b}R^2. \tag{20}$$

2.3 The Potential Cost Savings of Market-Based Policy Instruments

Employing our model of abatement costs of alternative policy instruments, the aggregate cost savings, $\tilde{\Delta}$, from using a cost-effective policy relative to a uniform emission rate standard is found simply by subtracting Equation (20) from (10):[10]

$$\tilde{\Delta}=C\left(\tilde{Q}\right)-C\left(Q^*\right)=\frac{Xa^2}{2b}\left(\nu\left(R^2+\mu\right)+\mu\right). \tag{21}$$

We can also express the cost savings in percentage terms by dividing Equation (21) by (10):

$$\%\tilde{\Delta}=1-\frac{R^2}{(1+\nu)\left(R^2+\mu\right)}. \tag{22}$$

Likewise, the aggregate cost savings, $\hat{\Delta}$, from using a cost-effective policy relative to a uniform percentage reduction standard is found by subtracting Equation (20) from (13):

$$\hat{\Delta}=C\left(\hat{Q}\right)-C\left(Q^*\right)=\frac{Xa^2R^2}{2b}\left(\nu\left(1+\mu\right)+\mu\right), \tag{23}$$

[10]Note that the cost savings approximation is likely to be acceptable for the policy-relevant range of emission reductions, but may not perform as well when aggregate reduction requirements are stringent enough to lead many sources to reduce all of their emissions under a market-based policy. This can occur in our current model since we do not impose a non-negativity constraint on emissions under a market-based allocation. It is easily seen by evaluating the cost savings at the extreme of 100 percent reductions (i.e., R=1). At 100 percent reductions the expressions suggest positive cost savings, although exact cost savings should be zero since every source would be required to emit zero emissions, facing the same costs under any policy. One could restrict the cost structure in order to avoid this limitation, for example by constraining the model parameters so that α_i/β_i is a constant equal to the marginal abatement cost at zero emissions. Given the choice between the aforementioned limitation and further model restrictions we chose to acknowledge the former rather than impose the latter.

or in percentage terms:

$$\%\hat{\Delta} = 1 - \frac{1}{(1+\nu)(1+\mu)}. \tag{24}$$

Equations (21)–(24) provide the basis for evaluating the potential cost savings from adopting a market-based policy instrument. Minimal information is required about the relevant set of sources facing environmental regulation: at most, aggregate production of the regulated industry, X; the emissions constraint, Q, which determines R; the mean and variance of the slope of the emissions demand function, b and V[β], which determine μ; and the mean and variance of baseline emissions intensity, a and V[α], which determine ν.

2.4 What Do the Expressions for Cost Savings Tell Us?

The first message from Equations (21)–(24) is that the cost savings of market-based policies relative to uniform performance standards increase in a straightforward manner as a function of greater abatement-cost heterogeneity.[11] This cost heterogeneity comes from two relevant sources. The first is heterogeneity in baseline emissions intensities, μ, which indicates how much abatement each source will have to do, depending on the policy in place. The second is heterogeneity in the slope of the abatement cost function, ν, which describes how fast each source's costs rise as additional reductions are sought. We find that each source of cost heterogeneity can have an effect independent of the other's presence, although the two interact when both are present. In particular, the effect of one type of heterogeneity on cost savings is amplified when the other type of heterogeneity is larger.

Further underscoring the importance of cost heterogeneity for instrument choice, we confirm that if there is no abatement-cost heterogeneity (that is, if both ν and μ are zero), then

there are no potential cost-savings associated with a market-based instrument ($\Delta = 0$), at least in terms of static cost-effectiveness.[12] The instruments would be "equivalent" since all sources would choose identical emissions intensities under a market-based instrument, which is precisely what a uniform performance policy requires. We also confirm, however, that the uniform performance standards cannot be *less* costly than a market-based policy instrument, since Equations (21) and (23) cannot be negative (because all of the variables and parameters are themselves non-negative). Note also that heterogeneity in source size (χ) does not enter the expressions, because the market-based policy does not offer any more flexibility along the size dimension than the performance standards, which themselves include an adjustment for size differences (on either a per unit of output or percentage basis). This result depends of course on the assumption that source size and the other heterogeneous variables (α and β) are independent.

Next, we turn specifically to Equations (21) and (23), which express the cost savings in dollar terms, and focus on the initial term multiplying the entirety of each of the expressions, $Xa^2/(2b)$. This term equals the aggregate cost of 100-percent emission reduction for a homogeneous set of sources. It serves to scale the expression to a degree appropriate for any particular environmental problem, depending on the size of the industry, baseline emissions, and the slope of control costs. The remaining variables in the expressions, μ, ν, and R, are dimensionless. When the cost savings are expressed in percentage terms, as in (22) and (24), only the dimensionless variables remain. Although expressing the cost savings in percentage terms has obvious appeal, one loses an overall sense of the importance of the choice at hand. Hence, both forms are useful.

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[11]The relevant derivations for this section are provided in the Appendix.

[12]As discussed in the introduction, there might still be advantages in terms of dynamic incentives for technological change and industry entry and exit decisions. In addition, we have not addressed any potential differences in administrative implementation costs.

Finally, we can consider how increases in the stringency of emissions reductions, R, interacts with cost heterogeneity to influence cost savings. As shown in the Appendix, the effect of more stringent reduction rates on absolute cost savings is amplified by higher degrees of cost heterogeneity.

These relationships between the cost savings of market-based policies and the various types and degrees of cost heterogeneity are illustrated in Figures 1 and 2. Since general results can be illustrated for cost savings measured in percentage terms, we focus on such measures. Figures 1 portrays the anticipated cost savings (in percentage terms, on the vertical axis) of employing a fully cost-effective market-based policy instrument instead of a uniform emission rate standard in the case of an aggregate emission reduction target of 50 percent. Increasing heterogeneity in the slope of marginal abatement costs is measured on the horizontal axis (in terms of the coefficient of variation of the slope of emissions demand, v). The general relationship between heterogeneity in marginal abatement cost function slope and cost savings are provided in the figure for four different degrees of heterogeneity in baseline emission rate (in terms of the respective coefficient of variation, μ). Figure 2 provides the analogous cost savings results relative to a uniform percentage reduction standard.

As can be seen in Figure 1, even in the absence of "slope heterogeneity," increasing heterogeneity in the baseline emission rate brings with it greater cost savings due to employing a market-based instrument (see the four vertical intercepts in the figure). Likewise, even when sources are identical in terms of their baseline emission rates ($\mu = 0$), percentage cost savings increase with greater degrees of heterogeneity in marginal abatement cost function slopes, v. Finally, note that there are "decreasing returns" to cost-effectiveness from greater degrees of cost heterogeneity. That is, relatively small degrees of both types of abatement-cost heterogeneity

result in market-based instruments enjoying significant advantages over their command-and-control counterparts. But there is a limit to

3. Application to Nitrogen Oxides Control

As an example of how these relationships can be employed, we apply the modeling framework developed above to the case of nitrogen oxides (NO_x) reduction in a group of northeastern states. When emitted, NO_x emissions and volatile organic compounds react in the presence of sunlight to form compounds that contribute to the formation of ground-level ozone. Ozone in the lower atmosphere can cause a variety of health problems because it damages lung tissue, reduces lung function, and adversely sensitizes the lungs to other irritants. Ozone tends to be a problem over broad regional areas, particularly in the eastern United States, where it can be transported by wind over hundreds of miles and across state boundaries. Through a multi-year effort known as the Ozone Transport Assessment Group (OTAG), the U.S. Environmental Protection Agency (EPA) worked with eastern states and the District of Columbia, private industry, and environmental advocacy groups to address ozone transport.

In 1998, EPA issued regulations requiring 22 eastern states and the District of Columbia to submit State Implementation Plans (SIP's) that address the regional transport of ground-level ozone through NOx reductions (U.S. Environmental Protection Agency 1998). Building on OTAG recommendations, EPA established NO_x budgets for each state, but gave states the flexibility to decide which electric utility boilers, large industrial boilers, and other sources should be required to reduce NO_x emissions, by how much, and the specific policies they must follow to meet the projected budgets (e.g., uniform emission rates, percentage reduction standards, or emissions trading). To determine the budgets, EPA chose a control level for large electric utility boilers based on a uniform emission rate standard of 0.15 lb/mmBtu (pounds of

NOx per million Btu of boiler heat input). For large industrial boilers and turbines (above 250 mmBtu/hr of heat input) EPA proposed a control level corresponding to a 60 percent reduction from an uncontrolled baseline.

We model several different emission control policies for large electric utility and industrial boilers in the relevant region, including: (1) a 0.15 lb/mmBtu uniform emission rate standard for utilities; (2) a uniform 60% emission reduction standard for industrial boilers; and (3) cost-effective allocations among utilities and industrial boilers, both individually and jointly.[13] The virtue of the approach we have developed is its analytic simplicity and its minimal information requirements; indeed all of the necessary data on control costs, emissions, and output are available as projections for 2007 (Pechan Associates 1997).[14] The necessary summary statistics for these data are given in Table II.

The results are provided in Table II and illustrated by the points labeled "NOx: Utilities" in Figure 1 and "NOx: Industrial point sources" in Figure 2. Thus, Figure 1 indicates that given the estimated degrees of heterogeneity of the two relevant cost function parameters, for electric utility NOx emissions our model predicts a best guess of 56% cost savings of employing a market-based policy instrument compared with a uniform emission rate standard. Although the policy comparisons we model are somewhat different, our results are in the same range as those of Krupnick and McConnell (1999), who employ a source-by-source linear programming approach to evaluate cost-effective NOx control and find cost savings for utilities of about 46%. For industrial point sources, which exhibit a much higher degree of heterogeneity, we estimate

[13]The application is similar to that in a study by Krupnick and McConnell (1999), who employ a linear programming approach to evaluate cost-effective NOx control.

[14] Costs for utilities are relative to an emission baseline consistent with meeting restrictions imposed by the 1990 Clean Air Act Amendments.

an 80% cost savings from the cost-effective allocation relative to the uniform percent reduction standard.

We also show the cost for each set of sources of employing the alternative type of performance standard (i.e., percent reductions for utilities and rate standards for industrial point sources). The results illustrate that percent reduction standards are less costly than uniform emission rate standards—unless of course there is no heterogeneity in baseline emission rates, in which case the two policies are identical. Finally, we analyze the cost savings associated with a jointly cost-effective allocation among utilities and point sources (e.g., trading allowed between utilities and point sources). Compared to the specific performance standards specified earlier, we find joint savings of 58% for a market based instrument. Compared to separate cost-effective allocations (e.g., trading within but not between utilities and point sources), however, we find that the jointly cost-effective allocation only offers a much smaller 3% decline in costs. This is attributable primarily to the fact that industrial sources make up only a small part of total emissions.

4. Conclusion

Policy makers and policy analysts in the environmental realm are frequently faced with the situation where it is unclear whether market-based instruments hold significant promise of reducing costs, relative to conventional command-and-control approaches. We have developed some rather simple rules-of-thumb that can be employed with relatively small amounts of information to estimate the potential cost savings that can be anticipated from designing and implementing market-based policy instruments. Because our analytical models are simple, yet capture key properties of pollution abatement cost functions, they can be used to predict potential cost savings through simple formulae.

The framework we have developed may be usefully applied to address a variety of other public policy questions. For example, what are the costs of maintaining one-size-fits-all environmental (and other) regulations across heterogeneous regions, such as the countries of Europe, the provinces of Canada, or the states of the United States? And what are the costs of uniform acreage control programs, such as in the European Union, when farms and farmers are highly heterogeneous?

The parsimonious set of intuitive rules-of-thumb we have developed can improve understanding of the importance of cost heterogeneity and its policy implications in real-world environmental and resource policy contexts. We hope that these simple formulae can aid policy analysts and policy makers in the early stages of exploring alternative policy instruments by helping them identify approaches that merit greater attention and more detailed analysis.

Table I. Variable Definitions and Units

Variable	Definition	Example of Units
a	average baseline emission intensity	(lb emissions)/(mmBtu output)
$\mathrm{V}[\alpha]$	variance of baseline emission intensities	$(\text{lb/mmBtu})^2$
b	average rate of change of emission intensity per unit change in emissions price	(lb/mmBtu)/($/lb)
$\mathrm{V}[\beta]$	variance of rate of change of emission intensity per unit change in emissions price	$(\text{lb/mmBtu})^2/(\$/\text{lb})^2$
X	aggregate output	mmBtu
$\bar{x}$	average output	mmBtu
p^*	emission tax or market-clearing price	$/lb
Q/X	aggregate (and average) emissions intensity constraint	lb/mmBtu

Table II. Application to Nitrogen Oxides Control in Northeastern United States

Variable	Electric Utilities	Industrial Point Sources	Joint Control
a (lb/mmbtu)	0.32	0.43	.34
$\mathrm{V}[\alpha]$ (lb/mmbtu)^2	0.010	0.16	.035
μ	0.098	0.83	0.31
b (lb/mmBtu)/($/lb)	0.15	0.38	.19
$\mathrm{V}[\beta]$ (lb/mmBtu)^2/($/lb)^2	0.015	.250	.060
ν	0.68	1.77	1.74
X (mmBtu)	9.85 x10^9	3.56 x10^8	1.02 x10^{10}
Q/X (lb/mmbtu)	0.15	0.17	.151
R	53%	60%	56%
$C(\tilde{Q})$ ($M)	2,220	290	—
$C(\hat{Q})$ ($M)	1,810	160	—
$C(Q^*)$ ($M)	980	30	990
p^* ($/lb)	1.16	0.69	1.02
$\tilde{\Delta}$ ($M)	1,240	260	—
$\%\tilde{\Delta}$	56%	89%	—
$\hat{\Delta}$ ($M)	830	130	—
$\%\hat{\Delta}$	46%	80%	—
Joint Control			
Uniform Standards ($M)	—	—	2,380
Cost-Effective Allocation ($M)	—	—	990
Difference in Uniform vs Cost-Effective ($M)	—	—	1,390
% Difference	—	—	58%
Difference in Cost Effective; Joint vs Separate ($M)	—	—	20
% Difference	—	—	3%

Figure 1. Cost Savings of Market-Based Policy Relative to Uniform Emission Rate Standard (50% reduction in aggregate emissions)

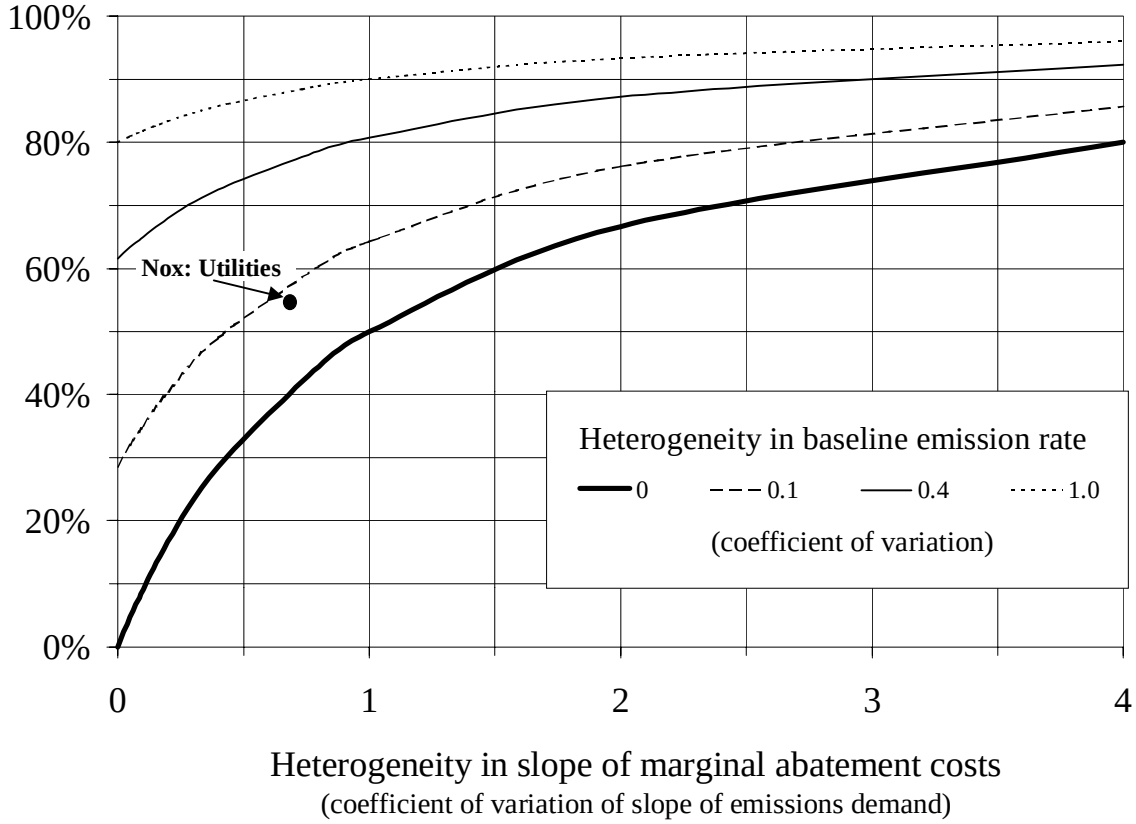
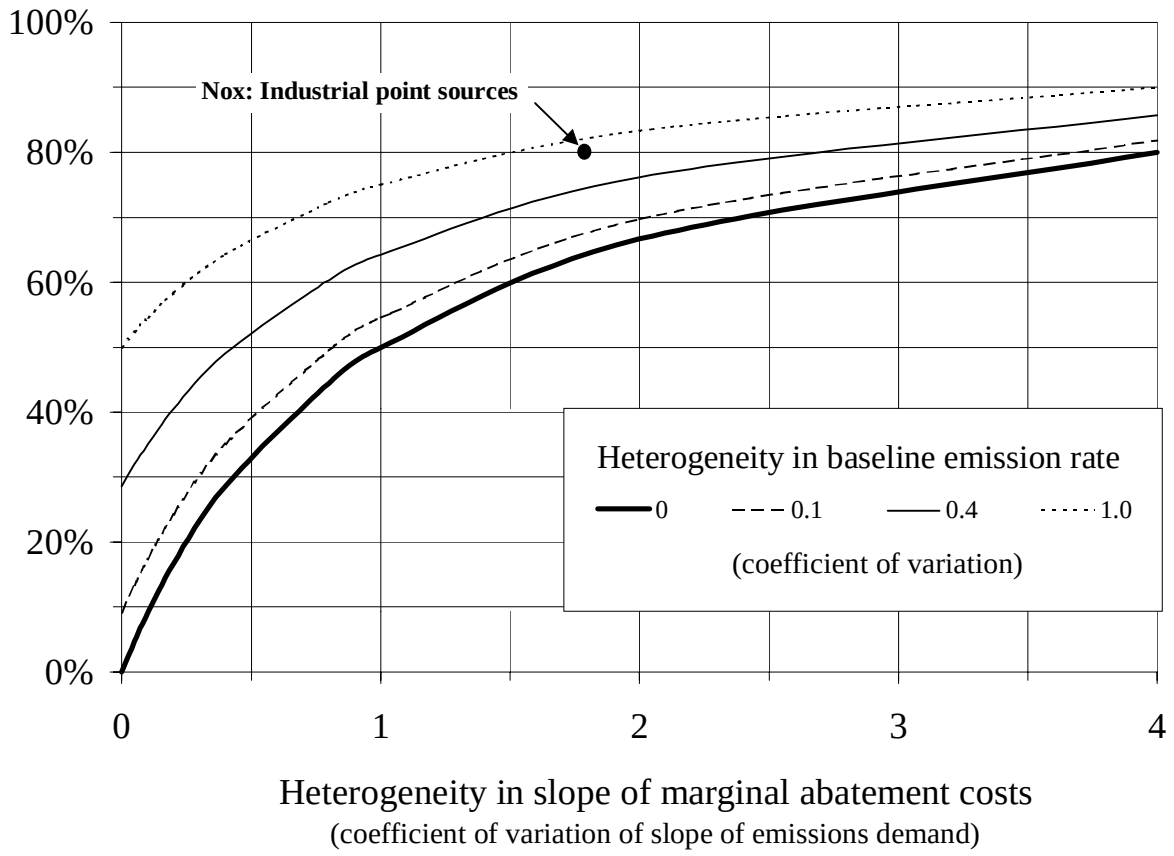


Figure 2. Cost Savings of Market-Based Policy Relative to Uniform Percent Reduction Standard



Appendix

Comparative statics on $\tilde{\Delta}$:

$$\left.\frac{d\tilde{\Delta}}{d\nu}\right|_b = \frac{Xa^2}{2b}\left(R^2+\mu\right) \tag{25}$$

$$\left.\frac{d\tilde{\Delta}}{d\mu}\right|_a = \frac{Xa^2}{2b}\left(1+\nu\right) \tag{26}$$

$$\left.\frac{d\tilde{\Delta}}{dR}\right|_a = \frac{Xa^2\nu}{2b}. \tag{27}$$

Comparative statics on $\%\tilde{\Delta}$:

$$\left.\frac{d\%\tilde{\Delta}}{d\nu}\right|_b = \frac{R^2}{\left(1+\nu\right)^2\left(R^2+\mu\right)} \tag{28}$$

$$\left.\frac{d\%\tilde{\Delta}}{d\mu}\right|_a = \frac{R^2}{\left(1+\nu\right)\left(R^2+\mu\right)^2} \tag{29}$$

$$\left.\frac{d\%\tilde{\Delta}}{dR}\right|_a = -\frac{2R\mu}{\left(1+\nu\right)\left(R^2+\mu\right)^2}. \tag{30}$$

Comparative statics on $\hat{\Delta}$:

$$\left.\frac{d\hat{\Delta}}{d\nu}\right|_b = \frac{Xa^2R^2}{2b}\left(1+\mu\right) \tag{31}$$

$$\left.\frac{d\hat{\Delta}}{d\mu}\right|_a = \frac{Xa^2R^2}{2b}\left(1+\nu\right) \tag{32}$$

$$\left.\frac{d\hat{\Delta}}{dR}\right|_a = \frac{Xa^2R}{b}\left(\nu\left(1+\mu\right)+\mu\right). \tag{33}$$

Comparative statics on $\%\hat{\Delta}$:

$$\left.\frac{d\%\hat{\Delta}}{d\nu}\right|_b = \frac{1}{\left(1+\nu\right)^2\left(1+\mu\right)} \tag{34}$$

$$\left.\frac{d\%\hat{\Delta}}{d\mu}\right|_a = \frac{1}{(1+\nu)(1+\mu)^2} \tag{35}$$

$$\left.\frac{d\%\hat{\Delta}}{dR}\right|_a = 0\,. \tag{36}$$

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