Market Mechanisms and Incentives: Applications to Environmental Policy

A Workshop Sponsored by the U.S. Environmental Protection Agency's National Center for Environmental Economics (NCEE) and National Center for Environmental Research (NCER)

Resources for the Future
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Market Mechanisms and Incentives: Applications to Environmental Policy

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1616 P Street, NW, Washington, DC 20036
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October 17th – 18th, 2006

October 17, 2006: Market Mechanisms in Environmental Policy

8:00 a.m. – 8:45 a.m.	**Registration**		
8:45 a.m. – 11:45 a.m.	**Session I: Brownfields and Land Issues** Session Moderator: **Robin Jenkins**, EPA, National Center for Environmental Economics		
		8:45 a.m. – 9:00 a.m.	Introductory Remarks: **Sven-Erik Kaiser**, EPA, Office of Brownfields Cleanup and Redevelopment
		9:00 a.m. – 9:30 a.m.	Environmental Liability and Redevelopment of Old Industrial Land **Hilary Sigman**, Rutgers University
		9:30 a.m. – 10:00 a.m.	Incentives for Brownfield Redevelopment: Model and Simulation **Peter Schwarz** and **Alex Hanning**, University of North Carolina at Charlotte
10:00 a.m. – 10:15 a.m.	**Break**		
		10:15 a.m. – 10:45 a.m.	Brownfield Redevelopment Under the Threat of Bankruptcy **Joel Corona**, EPA, Office of Water, and **Kathleen Segerson**, University of Connecticut
		10:45 a.m. – 11:00 a.m.	Discussant: **David Simpson**, EPA, National Center for Environmental Economics
		11:00 a.m. – 11:15 a.m.	Discussant: **Anna Alberini**, University of Maryland
		11:15 a.m. – 11:45 a.m.	Questions and Discussion
11:45 a.m. – 12:45 p.m.	**Lunch**		
12:45 p.m. – 2:45 p.m.	**Session II: New Designs for Incentive-Based Mechanisms for Controlling Air Pollution** Session Moderator: **Will Wheeler**, EPA, National Center for Economic Research		
		12:45 p.m. – 1:15 p.m.	Dynamic Adjustment to Incentive-Based Environmental Policy To Improve Efficiency and Performance **Dallas Burtraw, Danny Kahn,** and Karen Palmer, Resources for the Future

	1:15 p.m. – 1:45 p.m.	Output-Based Allocation of Emissions Permits for Mitigating Tax and Trade Interactions **Carolyn Fischer**, Resources for the Future
	1:45 p.m. – 2:00 p.m.	Discussant: **Ann Wolverton**, EPA, National Center for Environmental Economics
	2:00 p.m. – 2:15 p.m.	Discussant: **Arik Levinson**, Georgetown University
	2:15 p.m. – 2:45 p.m.	Questions and Discussion
2:45 p.m. – 3:00 p.m.	**Break**	
3:00 p.m. – 5:30 p.m.	**Session III: Mobile Sources** Session Moderator: **Elizabeth Kopits**, EPA, National Center for Environmental Economics	
	3:00 p.m. – 3:30 p.m.	Tradable Fuel Economy Credits: Competition and Oligopoly **Jonathan Rubin**, University of Maine; **Paul Leiby**, Environmental Sciences Division, Oak Ridge National Laboratory; and **David Greene**, Oak Ridge National Laboratory
	3:30 p.m. – 4:00 p.m.	Do Eco-Communication Strategies Reduce Energy Use and Emissions from Light Duty Vehicles? **Mario Teisl**, **Jonathan Rubin**, and **Caroline L. Noblet**, University of Maine
	4:00 p.m. – 4:30 p.m.	Vehicle Choices, Miles Driven, and Pollution Policies **Don Fullerton**, **Ye Feng**, and **Li Gan**, University of Texas at Austin
	4:30 p.m. – 4:45 p.m.	Discussant: **Ed Coe**, EPA, Office of Transportation and Air Quality
	4:45 p.m. – 5:00 p.m.	Discussant: **Winston Harrington**, Resources for the Future
	5:00 p.m. – 5:30 p.m.	Questions and Discussion
5:30 p.m.	**Adjournment**	

October 18, 2006:

8:45 a.m. – 9:15 a.m.	**Registration**	
9:15 a.m. – 12:20 p.m.	**Session IV: Air Issues** Session Moderator: **Elaine Frey**, EPA, National Center for Environmental Economics	
	9:15 a.m. – 9:45 a.m.	Testing for Dynamic Efficiency of the Sulfur Dioxide Allowance Market **Gloria Helfand**, **Michael Moore**, and **Yimin Liu**, University of Michigan
	9:45 a.m. – 10:05 a.m.	When To Pollute, When To Abate: Evidence on Intertemporal Use of Pollution Permits in the Los Angeles NOx Market **Michael Moore** and **Stephen P. Holland**, University of Michigan

10:05 a.m. – 10:20 a.m.	**Break**	
	10:20 a.m. – 10:50 a.m.	A Spatial Analysis of the Consequences of the SO_2 Trading Program **Ron Shadbegian**, University of Massachusetts at Dartmouth; Wayne Gray, Clark University; and Cynthia Morgan, EPA
	10:50 a.m. – 11:20 a.m.	Emissions Trading, Electricity Industry Restructuring, and Investment in Pollution Abatement **Meredith Fowlie**, University of Michigan
	11:20 a.m. – 11:35 a.m.	Discussant: **Sam Napolitano**, EPA, Clean Air Markets Division
	11:35 a.m. – 11:50 a.m.	Discussant: **Nat Keohane**, Yale University
	11:50 a.m. – 12:20 p.m.	Questions and Discussion
12:20 p.m. – 1:30 p.m.	**Lunch**	
1:30 p.m. – 4:35 p.m.	**Session V: Water Issues** Session Moderator: **Cynthia Morgan**, EPA, National Center for Environmental Economics	
	1:30 p.m. – 2:00 p.m.	An Experimental Exploration of Voluntary Mechanisms to Reduce Non-Point Source Water Pollution With a Background Threat of Regulation Jordan Suter, Cornell University, Kathleen Segerson, University of Connecticut, Christian Vossler, University of Tennessee, and Greg Poe, Cornell University
	2:00 p.m. – 2:30 p.m.	Choice Experiments to Assess Farmers' Willingness to Participate in a Water Quality Trading Market **Jeff Peterson**, Washington State University, and Sean Fox, John Leatherman, and Craig Smith, Kansas State University
2:30 p.m. – 2:45 p.m.	**Break**	
	2:45 p.m. – 3:15 p.m.	Incorporating Wetlands in Water Quality Trading Programs: Economic and Ecological Considerations **Hale Thurston** and Matthew Heberling, EPA, National Risk Management Research Laboratory, Cincinnati, Ohio
	3:15 p.m. – 3:35 p.m.	Designing Incentives for Private Maintenance and Restoration of Coastal Wetlands **Richard Kazmierczak** and **Walter Keithly**, Louisiana State University at Baton Rouge
	3:35 p.m. – 3:50 p.m.	Discussant: **Marc Ribaudo**, USDA, Economic Research Service
	3:50 p.m. – 4:05 p.m.	Discussant: **Jim Shortle**, Pennsylvania State University
	4:05 p.m. – 4:35 p.m.	Questions and Discussions
4:35 p.m. – 4:45 p.m.	**Final Remarks**	

4:45 p.m. **Adjournment**

Preliminary
Comments welcome

Environmental Liability and Redevelopment of Old Industrial Land

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ABSTRACT:

Many communities are concerned about the reuse of potentially contaminated land ("brownfields") and believe that environmental liability is a hindrance to redevelopment. However, with land price adjustments, liability might not impede the reuse of this land. Existing literature has found price reductions in response to liability, but few studies have looked for an effect on vacancies. This paper studies variations in state liability rules — specifically, strict liability and joint and several liability — that affect the level and distribution of expected private cleanup costs. It explores the effects of this variation on industrial land prices and vacancy rates and on reported brownfields in a panel of cities across the United States. In the estimated equations, joint and several liability reduces land prices and increases vacancy rates in central cities. Neither a price nor quantity effect is estimated from strict liability. The results suggest that liability is at least partly capitalized, but does still deter redevelopment.

Acknowledgements: I am grateful to Deepa Bhat and Leah Traub for excellent research assistance and to Anna Alberini, Howard Chang, Kathleen Segerson, David Simpson and workshop participants at MIT, Yale, Rutgers, the NBER, and the EPA for comments. This working paper was developed under STAR Research Assistance Agreement No. R831777 awarded by the U.S. Environmental Protection Agency. It has not been formally reviewed by the EPA. The views expressed in this document are solely those of the author and the EPA does not endorse any products or commercial services mentioned in this paper.

Many communities seek to encourage the redevelopment of sites that are idle or underused because of potential contamination (known as "brownfields"). Redevelopment of these sites is desirable because they are a disamenity and seen as a substitute for use of relatively pristine land (sometimes known as "greenfields"), which reduces open space and requires construction of new infrastructure. A survey by the U.S. Conference of Mayors (USCM) found about 25,000 brownfield sites in the 205 cities that responded (USCM, 2002).

Environmental liability — in particular, the threat of being compelled to pay for cleanup of contamination — is perceived as a significant barrier to redevelopment. The respondents to the USCM survey cited liability as second only to lack of cleanup funding as the major obstacle to redevelopment. In 2001, Congress passed the Small Business Liability Relief and Brownfields Revitalization Act, which funded and codified an existing EPA grant and loan program for cleanup of brownfields and included several provisions to reduce the presumed liability deterrent. Reuse of contaminated land remains an active issue for state and federal policy in the U.S. and abroad (Reisch, 2003; Grimski and Ferber, 2001).

Despite the perception of a problem, theoretical questions have been raised about the impact of liability in discouraging redevelopment (Boyd et al., 1996; Segerson, 1993, 1994). Much of the policy literature fails to consider real estate price adjustments in face of expected liability and thus may overstate the deterrent to redevelopment. Empirical questions about the role of liability also remain. Urban and industrial decline long predates modern environmental laws, so liability can be at best a partial explanation for underused industrial land. Previous literature has explored the effect of liability on prices but not on "mothballing" of land, with a few exceptions (McGrath, 2000; Schoenbaum, 2002).

In this paper, I use data on cities across the US to estimate the effects of variation in environmental liability on prices and vacancy rates of industrial land and on reported brownfield acreage. Most industrial land is potentially contaminated (Noonan and Vidich, 1992) and thus may be affected by liability, even if not formally designated as a brownfield; however, the brownfield designation may also apply to land contaminated by other uses.[1] The variation in liability comes from differences in state liability regimes, including whether they rely on strict liability and on joint and several liability. As explained below, these regimes affect the level and the distribution of expected private cleanup costs. States adopted and rescinded both forms of liability in the period in question, facilitating a panel data analysis. In addition, the effects of liability laws are compared across cities that differ in the likelihood of contamination to introduce intrastate as well as interstate variation.

This paper builds on the existing empirical literature in a few ways. First, it focuses on vacancy of industrial land and reported brownfield acreage, variables of policy interest. It is the first study to look at these quantity measures that does not use spatial heterogeneity in historical contamination as its explanatory variable. Second, it analyzes the effects of alternative liability rules and thus provides direct information on plausible policy reforms: complete elimination of liability is unlikely (and a history of contamination cannot be reversed), but some states have eliminated joint and several liability, and the U.S. federal government and many European countries have moved to restrict it. Third, it studies both urban and suburban data and thus provides some insight into not just the deterrent effect on redevelopment, but also substitution of greenfields.

I find a negative effect of joint and several liability on industrial real estate prices in central cities, with a reduction in prices of 14%, and a positive effect of joint and several liability on industrial vacancy rates, which is also confined to central cities. One cannot reject no effect of strict liability on either prices or vacancy rates. Tests do not find evidence of policy endogeneity for either the price or vacancy equations, lending support to the estimated coefficients. The results are inconclusive on the question of greenfield substitution.

The paper also analyzes the USCM survey, the one national data set on reported brownfields acreage. The survey has only been conducted over a limited time and does not standardize the definition of brownfields. However, the results provide some validation of the results for vacant industrial land. Reported brownfield acreage is higher with joint and several liability, but not with strict liability.

[1] Nonindustrial land, especially public facilities and commercial land, may account for 30% of reported brownfields (Wernsted et al., 2004).

The outline of the paper is as follows. The next section discusses reasons that liability may deter redevelopment and previous theoretical and empirical research in more detail. It also advances hypotheses about the effects of alternative liability rules. Section 2 describes the data on state policies, real estate markets, and other economic conditions merged for the analyses. Section 3 presents panel data estimates of equations for industrial land prices and vacancy rates and tests for the endogeneity of public policies. Section 4 describes the data set that was assembled around the four years of the USCM survey and results from equations estimated on these data. A final section briefly concludes with policy implications.

1 Liability as a deterrent to redevelopment

Under the federal Superfund and most state programs, liability for cleanup of contaminated sites may be imposed on a number of parties, including past and present owners of the site, as well as parties that contributed or transported contaminants to the site.[2] The purchaser of land bears the risk of liability should the site turn out to be contaminated. In addition, the original owner may not find its liability eliminated by the sale, given the inclusion of past landowners among the liable parties. This section discusses studies of the effects of liability on redevelopment and then considers their implications for specific liability rules.

1.1 Previous research on liability and redevelopment

The previous literature suggests four reasons that liability might not just lower land prices, "capitalizing" liability, but also deter redevelopment.

First, sellers of land and potential buyers may have asymmetric information about the level of contamination and the nature of the required cleanup. As Boyd et al. (1996) and Segerson (1994)

[2]Since 1986, the federal Superfund program has allowed an "innocent landowner" defense, which exempts purchasers who did not know the parcel was contaminated, made "all appropriate inquiry," and exercised due care once contamination was discovered. However, courts have applied various criteria for allowing this defense and in practice have rarely found it applicable. The Brownfields Act of 2001 clarifies these concepts (in particular, regulations issued in 2005 define "all appropriate inquiry") and may increase the frequency and reliability of this defense.

argue, the resulting adverse selection may be a source of underuse of old industrial land. Although insurance for buyers' cleanup costs has become increasingly available, this market too is likely to be imperfect.

Second, Boyd et al. examine what they call "imperfect detection," in which the government (and potentially even the owner) does not know about contamination until redevelopment, and "imperfect enforcement," in which the government does not enforce cleanup liability for idle sites. In these circumstances, the sale of the property or requests for development permits may cause the owner to bear cleanup costs it could otherwise escape. If the cost of cleanup exceeds the value of the site clean, the property may go undeveloped.

Third, Segerson (1993) explores the effects of the "judgment proof problem," the possibility that parties may escape full liability through bankruptcy. In Segerson's model, without judgment proof parties, sales will be efficient regardless of whether the liability is transferred, i.e., whether it continues to reside with the seller or is partly or fully taken by the buyer. But with judgment proof parties, the extent of this shifting (and thus liability rules and rules on disclosure) affect the efficiency of the outcome. Segerson (1994) applies her earlier analysis to the incentives to clean up contamination before sale. Although Segerson shows that the effect of liability on sales is theoretically ambiguous, the legal rules in place largely shift liability to buyers, who are likely to have deeper pockets than current owners. Thus, a deterrent effect seems the likely implication of her model in practice.

Fourth, Chang and Sigman (2005) identify several deterrent effects that derive specifically from joint and several liability. Joint and several liability allows the government to hold any party liable for all of the cleanup costs regardless of its share of responsibility; this party may then sue any remaining liable parties for their share. Chang and Sigman discuss four different effects, all of which result from the increase in the number of defendants with sale of the property. For example, at sites with multiple liable parties, a sale may shift some third-party liability to the buyer. In addition, the buyer and seller may have collectively greater expected liability than the seller alone when the outcomes of the government's potential lawsuits are imperfectly correlated among the different liable parties. Thus, joint and several liability in particular may be a culprit in any deterrent effect from liability.

Empirical research. A few previous studies have explored empirical determinants of redevelopment.[3] McGrath (2000) examines the sales prices and likelihood of redevelopment of industrial parcels in Chicago as a function of the probability of contamination, which he derives from historical land use. McGrath finds a price reduction of about $1 million per acre and, comparing this value to typical cleanup costs, suggests that the costs are fully (or even over-) capitalized. He also compares sites that sold for new industrial uses with sites that sold for current use and finds evidence that redevelopment was discouraged. However, this definition of redevelopment is narrow: most policy-makers are concerned about the "mothballing" of land, rather than the question of change in use. McGrath's study conditions on a transaction taking place and thus cannot address the broader question.

Schoenbaum (2002) provides the most rigorous previous examination of land vacancy. She constructs a history of land use in an industrially-zoned area in Baltimore. Categorizing some land as brownfields in 1963 and in 1999, she finds no evidence that either status affects the likelihood of vacancy in 1999. However, identification of the brownfield effect is potentially confounded with spatial heterogeneity; parcels with geographic advantages (for example, proximity to a highway) may be more intensively used and thus be both more likely to be brownfields and to be used again later. This concern is supported by the positive association between land values and brownfield status in her study.

Other studies focus on prices only. Jackson (2002) examines the price effect of known contamination and its cleanup on industrial land prices in Southern California and surveys other studies of the effects of contamination on land prices. These studies show price responses; however, they do not indicate the extent of capitalization or the effect on redevelopment.

Finally, two recent studies use stated-preference analysis to explore incentives to promote rede-

[3] In addition to statistical analyses, case studies include Zabel (2003), Nijkamp et al. (2002) and Urban Institute et al. (1997).

velopment. Alberini et al. (2005) surveyed European developers on the impact of liability reduction, regulatory relief (improved speed and flexibility in approving cleanup), and direct subsidies to the developer. They find liability relief is worth 21% of the value of the median development project. Wernsted et al. (2006) surveyed land developers in the United States. Using a conjoint choice analysis, they find that protection from third-party lawsuits is worth 22% of the return on investment at the hypothetical site and cleanup liability protection is worth 16%. These stated preference studies, however, cannot diagnose whether liability causes only price adjustments or also has an effect on quantities.

1.2 Effects of alternative liability rules

In the empirical analysis, variation in the extent of liability derives from the rules used to impose liability. In particular, the empirical analysis focuses on two dimensions of liability rules: whether liability is strict and whether it is joint and several. In this section, I discuss hypotheses about the relationship of these rules to redevelopment.

Strict liability means that any action that causes contamination may give rise to liability; by contrast, negligence (or "at fault") rules trigger liability only if precaution falls below some legal standard of care. Strict liability should increase expected private cleanup costs by expanding the set of sites at which private parties may be held liable. Under a negligence rule, parties will only find themselves liable only if they fail to exercise the legal standard of care (however the state defines this concept) in avoiding or cleaning up contamination. Under a strict liability rule, the government may also find it less costly to bring suits because its information requirements are lower, reinforcing the incentives from its higher expected awards.

Earlier empirical research supports this view. Previous papers find evidence of higher precaution with strict liability — reduced spills (Alberini and Austin, 1999b, 2002) and fewer violations of hazardous waste laws (Stafford, 2003).[4] These results are consistent with expected liability

[4]Such higher precaution suggests legal standards of care below the social optimum (Cooter and Ulen, 1988; Tietenberg, 1989). However, expected cleanup costs could be higher with strict liability even if it does not elicit greater precaution (as would be the case if legal standards of care are optimal).

costs that are higher with strict liability.

Joint and several liability may raise expected liability for developers for several reasons. As mentioned above, Chang and Sigman (2005) discuss ways that the increase in the number of liable parties under joint and several liability creates disincentives for sale when all parties are solvent. In addition, joint and several liability obliges private parties to pick up “orphan shares,” costs attributable to parties that have gone bankrupt; these costs would be paid by the government under non-joint, “several only,” liability. Probst et al. (1995) estimate a 14% average orphan share at federal Superfund sites (excluding entirely orphan sites), so these costs may be substantial.

2 Data

Data from several sources were combined to yield a panel on real estate markets, liability regimes, and economic conditions across cities.

2.1 State liability rules

All U.S. states have “superfund” programs that address liability and funding for cleanup of contaminated sites not covered under the federal Superfund program or the federal Resource Conservation and Recovery Act (RCRA).[5] States vary in the nature of the liability rules they apply.

Landowners and other parties face two liability regimes, the regime in their state and the federal law. However, state liability, designed to capture sites neglected by the federal government, may be the relevant liability threat for run-of-the-mill industrial sites. These sites do not have the large-scale contamination that would qualify them for the federal program. When cleanup is undertaken under state programs, federal officials almost never intervene and developers do not ask for federal officials to sign off on cleanup plans (Boyle, 2005).

The longest history of these policies is available from a series of approximately biennial studies

[5]Superfund addresses inactive contaminated sites, whereas RCRA’s Corrective Action is responsible for sites with active hazardous waste management.

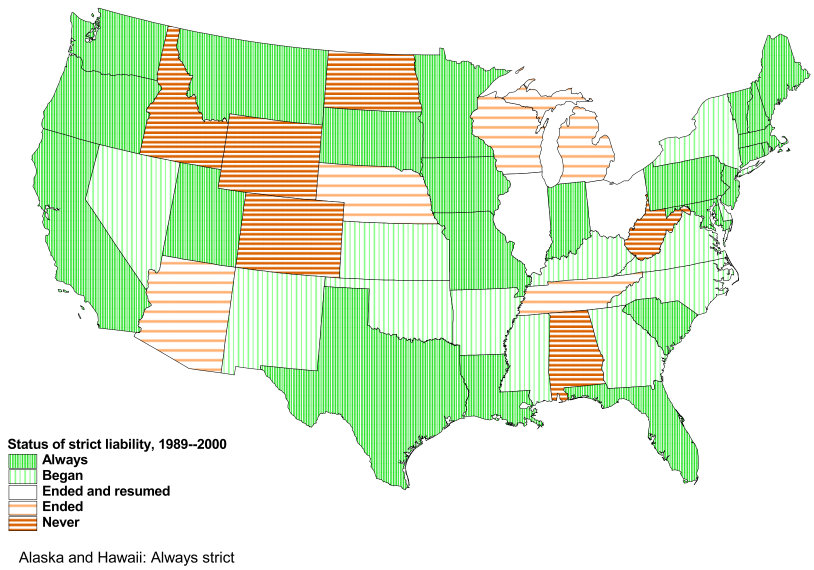

Figure 1: Reliance on strict liability, 1989–2000

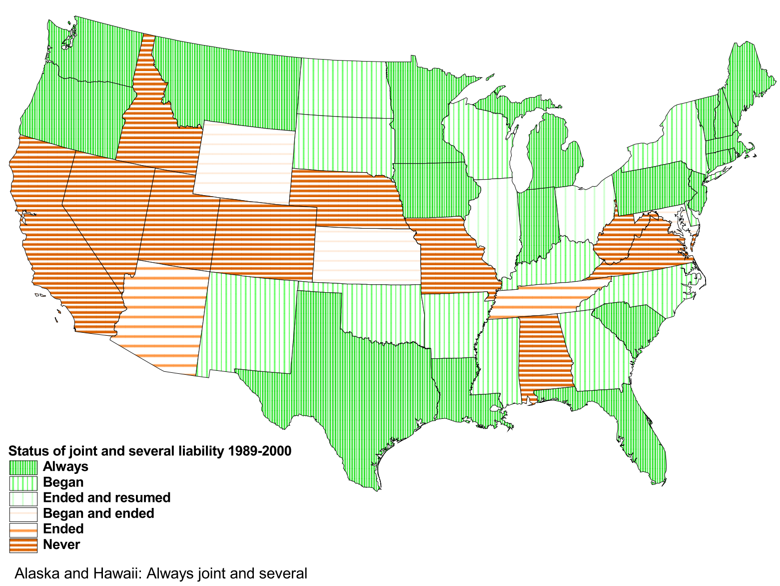

Figure 2: Reliance on joint and several liability, 1989–2000

from 1989 through 2000 by the Environmental Law Institute (ELI).[6] ELI surveys the state for its policy and says it captures not just the state's statute, but its current interpretation by the government. Figure 1 reports states that had strict liability and those that did not throughout the period, as well as states that switched from one regime to another. Figure 2 reports the same data for joint and several liability. For both liability regimes, states switched both to and from the rules within the period. The majority of transitions are permanent, at least as far as the data extend. Correlation between strict and joint and several liability is imperfect and not all transitions occurred in tandem.

The policies change as a result of legislative, judicial, and administrative decisions. A number of legislatures enacted "tort reform" before and during this period (Campbell, Kessler, and Shepherd, 1998). Abolishing or severely restricting joint and several liability has been a common component of these reforms, although environmental liability is often specifically excluded (American Tort Reform Association, 2005). Some of the shifts reported by ELI appear to be related to this wave of legislation and the judicial reaction. For example, a 1995 Illinois law barred joint and several liability, but the Illinois Supreme Court reinstated it in 1997; the hiatus in joint and several liability appears in the ELI data. In other instances, the policies are administrative. For example, Tennessee reports dropping joint and several liability in 1990, before a 1992 law passed.[7]

2.2 Land data

The Society of Industrial and Office Realtors' (SIOR) annual *Comparative Statistics of Industrial and Office Real Estate Markets* has data for many U.S. cities on prices of industrial real estate and vacancies. These data are available annually beginning in the early 1980s. The SIOR reports the expert opinion of local realtors rather than transaction data. Reliance on experts may add noise

[6] The years of the data are 1989, 1990, 1991, 1993, 1995, 1997, 2000 (see Pendergrass, 2001). For the econometrics, continuity in liability rules is assumed for intervening years (1992, 1994, 1996, 1998-99) when no change is reported. When the reports indicate a change, liability regime variables are missing for intervening years.

[7] Good and Richards (2004) believe that some of the apparent time-series variation is spurious, resulting from inconsistencies in responses to questionnaires across years, and propose to use statutory data only. However, this approach risks missing genuine policy shifts; environmental enforcement divisions may set a policy of not availing themselves of privileges the law affords. In any event, the measurement error from inconsistent responses to surveys should introduce a conservative bias to the empirical results.

because of the influence of respondents' impressions, but may also reduce the noise in price data that a few large sales might have generated in some of the smaller urban areas.

For many cities, the SIOR data provide separate central city and suburban price and vacancy rates. Suburban sites may be less likely to be contaminated than urban sites and thus provide a comparison group.[8] In addition, a frequent argument for brownfield redevelopment is that firms would otherwise substitute suburban for urban sites. The suburban data permit a direct test of this hypothesis, at least to the extent that the substitution would be toward suburban sites within the same metropolitan area.

Industrial land was chosen to represent land potentially affected by liability for several reasons. First, data are available for many cities over a long period. Second, almost all old industrial sites have some "environmental issues." Noonan and Vidich (1992) surveyed environmental engineering firms to determine the probability of contamination for different land uses. They report very high probabilities for all the industrial uses: land used for "heavy industrial manufacturing" has a probability of contamination of 88% and "light industrial manufacturing" and "industrial parks" have 75% probabilities. Thus, it is highly likely that land zoned as industrial is contaminated, especially in center cities where it may have seen extensive previous use. Third, liability might cause a general cooling of industrial real estate markets, which could be more costly than its effect on a few high-profile brownfields. In particular, adverse selection might be a problem for the market as a whole, but not for sites with well-established contamination. The disadvantage of studying industrial land is that land with other sources of contamination, such as ubiquitous brownfields from abandoned auto repair shops, falls outside the analysis. For this reason, designated brownfields are considered later.

Table 1 reports mean prices and vacancy rates for industrial land in central city and suburban locations. Prices are substantially higher and vacancy rates lower in the suburbs. The table also distinguishes both variables for observations with and without joint and several liability. Center city and suburban prices are lower and vacancy rates higher with joint and several liability; the

[8] A number of suburban observations have been discarded, however, because the areas in question span more than one state, so the liability regime is ill-defined.

disparities are smaller between suburban values than urban values, consistent with an effect that depends on the likelihood of contamination.

Tables 2 and 3 provide a "difference in difference" analysis of changes in joint and several liability. For cities with both urban and suburban data, the tables present the ratio of urban to suburban land prices (Table 2) and urban to suburban vacancy rates (Table 3) in 1989 and 2000. The cities are divided according to whether their states always used (or did not use) joint and several liability or switched regimes "permanently" during the study period. For all groups of cities in Table 2, urban prices fell relative to suburban prices over the time period. However, the relative fall in urban prices was substantially smaller for the group that left joint and several liability than those that remained. Similarly, among cities that initially did not have joint and several liability, cities that began it had a greater relative reduction than those that did not. A similar story emerges in Table 3. Vacancy rates in the center fell relative to the suburbs in cities where joint and several liability ended, whereas cities that maintained joint and several liability saw a relative increase in vacancies. Cities that began joint and several liability had an increase in their relative urban vacancy rate, whereas those that never had it experienced a fall. The differences, therefore, are consistent with a reduction in land prices and increase in vacancy rates from joint and several liability, although none of the differences are close to statistically significant. Sample sizes are small, especially in the transition categories.

2.3 Other explanatory variables

Other explanatory variables reflect economic conditions in the city, government services and taxes, influences on expected cleanup costs other than liability rules, and descriptions of other state environmental policies.

For economic conditions, the equations include the unemployment rate, manufacturing employment, and city population. The Bureau of Labor Statistics (BLS) provides data by city on unemployment rates and manufacturing employment; Wheaton and Torto (1990) suggest that the latter plays an important role in industrial real estate demand. Table 1 reports that the mean of this

Table 1: Means of variables used in equations, by joint and several liability

	All obs		**Without J&S**		**With J&S**	
	Mean	**St dev**	**Mean**	**St dev**	**Mean**	**St dev**
Price — center city (2000$/sq ft)	26.1	17.2	29.6	19.1	23.9	15.5
Price — suburb (2000$/sq ft)	29.5	15.5	34.8	20.7	26.9	11.2
Vacancy rate – center city	9.89	8.08	9.18	6.87	10.3	8.73
Vacancy rate – suburb	8.06	5.40	7.94	5.73	8.12	5.22
Strict liability	.829	.377	.596	–	.956	–
Joint and several liability	.645	.479	0	–	1	–
Metropolitan population (million)	3.48	11.0	2.05	2.75	4.23	13.4
Unemployment rate (%)	5.00	2.23	5.26	2.79	4.87	1.87
Manufacturing employment (thous)	105	138	131	190	91.1	96.0
Highway density	.264	.087	.265	.089	.263	.086
Real estate taxes (2000$/sq ft)	.800	2.35	.605	1.46	.889	2.66
State superfund lawyers per million people	.772	.814	.757	.825	.781	.809
League of Conservation Voters score	45.9	18.0	41.5	13.6	48.3	19.5
Abatement cost index	1.03	.347	.945	.256	1.07	.380
Contaminated sites/ sq mile	.086	.084	.075	.078	.091	.086
Historical manufacturing workers/ sq mile	131	348	34.2	79.9	183	419

Table 2: Urban-suburban price ratios by liability regime and transitions, 1989 and 2000

	Price ratios		**Difference**	
	1989	**2000**	**Mean**	**(St. error)**
Cities with joint and several liability	.943	.797	-.146	(.062)
	[30]	[29]		
Cities ending joint and several liability	.855	.778	-.077	(.191)
	[5]	[9]		
Cities without joint and several liability	.922	.815	-.108	(.083)
	[12]	[14]		
Cities beginning joint and several liability	.971	.833	-.138	(.137)
	[9]	[9]		

Note: Numbers in square brackets are the number of urban/suburban pairs with data.

Table 3: Urban-suburban vacancy rate ratios by liability regime and transitions, 1989 and 2000

	Vacancy rate ratios		Difference	
	1989	**2000**	**Mean**	**(St. error)**
Cities with joint and several liability	1.15	1.42	.26	(.34)
	[26]	[25]		
Cities ending joint and several liability	1.72	1.12	-.66	(.56)
	[4]	[4]		
Cities without joint and several liability	1.68	1.43	-.25	(.40)
	[12]	[16]		
Cities beginning joint and several liability	1.75	2.83	1.08	(.97)
	[10]	[11]		

Note: Numbers in square brackets are the number of urban/suburban pairs with data.

variable is much larger in states without joint and several liability than in states with it; however, the difference is almost entirely driven by the upper tail and the medians are similar. The population for the metropolitan area, from the Census, is also included; the entire metropolitan area is used, regardless of whether the area is center city or suburbs.

The services provided and taxes collected by the city also contribute to real estate demand. In particular, surveys find that access to transportation is a major determinant of firms' location choices (Robertson, 1999). The Federal Highway Administration provides annual city-level data on highway miles that can be used to calculate a time-varying measure of highway density for each urban area. For taxes, SIOR provides an estimate of real estate taxes per square foot beginning in 1994.

Some additional sources of variation in the expected costs of liability can be captured for the analysis. The likelihood of contamination varies with past industrial land use. Fixed effects will remove the levels of these effects. However, for interaction terms, the equations use two different measures of the legacy of contamination. First, EPA's inventories of suspected and confirmed contaminated sites have been aggregated to the city level to create a measure of the geographic density of contaminated sites.[9] A second measure of the legacy of contamination is historical

[9]The variable includes both sites in the Comprehensive Environmental Response, Compensation, and Liability Information System (CERCLIS) and sites that have been moved to the No Further Remedial Action Planned (NFRAP) list. Both inventories have a field for SMSA, but it is rarely filled in, so the variable aggregates sites to the SMSA level

manufacturing employment. Data are available each decade from 1940 through 1970; these four decennial values have been averaged and divided by land area to create a measure of the spatial intensity of manufacturing in the past. Observations in northern New Jersey have the highest density, whereas Reno, NV has the lowest. Intrastate variation is present; for example, California's coastal cities have substantially higher values than cities in its Central Valley.

Another source of variation in the expected costs of liability is the aggressiveness with which a state pursues cleanup. Alberini and Austin (2002) capture this variation with the number of lawyers (full-time equivalent) working for the state on contamination, data which are available from ELI. As Table 1 reports, state have nearly identical staffing of their contaminated site programs regardless of whether they have joint and several liability.

Finally, explanatory variables are included to capture broad environmental policy stringency at the state level. This heterogeneity is potentially correlated with liability regimes and thus important to include in the equations. I use two variables: a measure of state environmental sentiment and a measure of manufacturing pollution abatement costs. The measure of environmental sentiment in the state is the average League of Conservation Voters (LCV) score for the House delegation of the state. The LCV score (which ranges from 0 to 100) represents the share of a legislator's votes on selected measures that the LCV considers pro-environment. As a measure of environmental sentiment, LCV scores have the virtue of varying over time and of perhaps reflecting the position of the median voter in the state (in contrast, for example, to environmental group membership, which focuses on the upper tail). I use House rather than Senate scores because the House scores usually average more individual legislators' data than Senate scores, reducing noise, and also can adjust more rapidly to changes in sentiment because of the potential for faster turnover in the House. As Table 1 reports, this variable is higher in states with joint and several liability, suggesting that it may be seen as the "greener" choice.

The measure of regulatory stringency is Levinson's industry-weighted abatement cost index (Levinson, 2000). Levinson adjusts the data from the U.S. Census survey on Pollution Abatement

by county, which is almost always reported in the Superfund data. County-level historical manufacturing data were also aggregated to the SMSA level.

Costs and Expenditures (PACE) for the two-digit industry composition in the state. The resulting index has the advantage of varying over time and capturing not just legislative differences between states but also differences in monitoring and enforcement. A major disadvantage is that it ends in 1994 when the Census stopped conducting its annual survey. The series is linearly extrapolated for later years. The index differs very little between observations with and without joint and several liability, which is somewhat surprising given the higher LCV scores in joint and several states.

3 Econometric analysis

In this section, I present estimates of the relationship between liability rules and two real estate market outcomes: prices and vacancy rates. The first two subsections use fixed effects estimators to capture unobserved heterogeneity, but otherwise assume exogeneity of the policy regime. In the last subsection, I discuss a test of exogeneity of liability regimes.

The equations are estimated only on data from 1989 through 2000. Because it is unclear when in the year the ELI survey describes, I use a one year lag to assure that the variable has the value relevant when planning for any transaction occurred. Thus, the remaining observations begin in 1990, which is convenient because it is also the first year in which manufacturing employment and highway density are available and avoids some complications from redefinition of urban areas between decennial Censuses.

For both price and vacancy rates, the estimated equations have the form

$$Log(p_{it}) = f(L_{it}, E_{it}, G_{it}, S_{it}) + \alpha_i + \beta_t + u_{it},$$

where variables are as follows: p_{it} is the price (or later the vacancy rates); L_{it} is a vector of state liability rules; E_{it} are economic conditions, such as unemployment and population; G_{it} are government variables (highway density and real estate taxes); and S_{it} are measures of state environmental policy. The equations also include a city fixed effect, α_i; Hausman tests reject random effects. Year effects, β_t, capture changes in interest rates and other national real estate trends. A log-log function form is used to allow variables that reflect the scale of activity, such as population, to interact multiplicatively with other variables.

The error is allowed to have an AR(1) structure within a panel,

$$u_{it} = \rho u_{it-1} + \varepsilon_{it}.$$

This error structure may capture not only the gradual change in unobservable characteristics, but also some tendency for slow adjustment in the opinions of the realtors who report data. The test suggested by Wooldridge (2002, p. 275) for autocorrelation in fixed effects models strongly rejects the hypotheses of no autocorrelation for both sets of equations. Estimates of ρ are large, as reported in the tables.

3.1 Panel data analysis: Prices

Table 4 presents estimates of the relationship between liability rules and prices. Four different equations are shown in Table 4. The first three equations restrict the sample to center city data only. This restriction is intended to focus attention on properties where some contamination is likely. The third equation in Table 4 includes all data from the SIOR, including both center city and suburban data.

I discuss the coefficients on the liability variables first and then discuss the other covariates.

Liability variables. In the first equation, joint and several liability has a statistically significant negative effect on prices. Prices are 14% lower (based on the coefficient of -.146) with joint and several liability, suggesting substantial capitalization of expected private cleanup costs. This price reduction is similar to the value of cleanup liability relief (16% of site value) in the stated preference study by Wernsted et al. (2004).

Strict liability is not observed to have an effect on prices. In column (1), the coefficient on strict liability is positive, but small in magnitude and statistically insignificant. The failure to find effects

Table 4: Panel estimates for price with fixed effects and AR(1) disturbances

	Dependent variable: Log(Price)			
	Center only			All obs
State liability rules				
Strict liability	.106	.331	-.069	.053
	(.068)	(.193)	(.145)	(.057)
Strict * Log(Site density)	–	.060	–	–
		(.052)		
Strict * Log(Old manuf emp)	–	–	.069	–
			(.045)	
Strict * Center city	–	–	–	.038
				(.084)
Joint and several liability	-.146	-.400	.095	-.044
	(.074)	(.242)	(.155)	(.064)
Joint and several * Log(Site density)	–	-.080	–	–
		(.073)		
Joint and several * Log(Old manuf)	–	–	-.085	–
			(.049)	
Joint and several * Center city	–	–	–	-.104
				(.094)
Other variables				
Log(City population)	.198	-.217	-.210	-.048
	(.034)	(.135)	(.142)	(.079)
Log(Unemployment rate)	.045	.006	.014	.012
	(.058)	(.056)	(.057)	(.038)
Log(Manufacturing employment)	-.038	.182	.186	.069
	(.136)	(.163)	(.168)	(.094)
Log(Highway density)	-.114	-.091	-.109	-.045
	(.092)	(.088)	(.091)	(.048)
Log(Real estate taxes)	.027	.036	.038	.033
	(.024)	(.023)	(.023)	(.018)
Log(State superfund lawyers)	.045	.059	.062	.050
	(.023)	(.023)	(.023)	(.015)
Log(LCV score)	.001	.034	.033	.025
	(.053)	(.051)	(.052)	(.035)
Log(Abatement cost index)	.050	.010	.007	.020
	(.079)	(.079)	(.081)	(.050)
F-test for strict & strict interaction		2.30	2.88	1.46
p-value		.101	.057	.233
F-test for J&S & J&S interaction		2.58	3.31	2.53
p-value		.077	.038	.081
ρ for AR(1) process	.45	.49	.49	.46
Number of cities	85	82	81	177
Number of observations	537	522	512	1195

Notes: Standard errors in parentheses.
Not shown: year dummies, dummy for missing highway observations, missing tax, and missing manufacturing employment.

of strict liability, here and below, may indicate that this form of liability is in fact no more stringent than the alternative of negligence rules. If the standard of care required to avoid negligence is high relative to the distribution of care actually taken, negligence rules protect few parties from liability.[10]

The next two equations in Table 4 explores interactions of legal regimes with the intensity of contamination, introducing intra-state variation into the identification of the effects. In column (2), the log of the density of hazardous waste sites in the metropolitan area is the measure of intensity of contamination. The point estimate on the interaction with joint and several liability is negative, consistent with the hypothesis that joint and several liability has a larger negative effect on prices the more likely property is to be contaminated. However, the two joint and several liability variables are jointly significant only at 10% and the interaction term is not individually significant. For strict liability, the effects remain insignificant and opposite in sign.

In column (3), the intensity of contamination is measured by the geographic density of historical (1940–1970) manufacturing employment. The interaction of this variable with joint and several liability also produces a negative coefficient as expected. Although the level of the joint and several liability is positive, the net effect at the sample median manufacturing employment is a reduction in price of 14%, which is statistically significant and very close in magnitude to the main effect in column (1). The two joint and several liability coefficients are jointly statistically significant at 5%. Thus, the results are consistent with a stronger negative association where contamination is more likely.[11]

In the fourth equation, suburban observations are added. The liability rules are interacted with a dummy for center city location to allow differentiated effects. The point estimates suggest a negative effect of joint and several liability overall that is strongest in city centers. However,

[10] These results could be consistent with earlier studies that find effects of strict liability on current precautions (Alberini and Austin, 1999b, 2002; Stafford, 2003). The analysis here compares the distribution of past precaution with the current standard of care. Current precaution levels may be enough higher that the standard of care is relevant.

[11] Another possible interaction with the liability rules is with unemployment rates as a measure of overall economic climate. In boom times, demand for land is high enough that even with imperfect detection/enforcement, it will be worth developing land. Thus the effects of liability would be stronger when unemployment is higher. However, these interactions were neither statistically significant nor consistent in sign across specifications.

neither coefficient is individually statistically significant and the two joint and several coefficient are jointly significant only at 10%. The sum of the two effects (the net effect in center cities) is similar in magnitude to the center city effect in the first equation. With negative point estimates for suburban areas, the results do not suggest substitution of suburban sites for central sites within the same metropolitan area in response to joint and several liability. Effects of strict liability remain statistically insignificant, small in magnitude, and perverse in sign.

Timing issues are a concern for these and other equations: a prospective property developer will care about expected current and future liability. Current liability rules will be a component of these expectations both for its direct effect (cleanup is likely to be required immediately before development can begin) and also for its indications about the future. However, unobserved expectations about the future may also play a role. If rules change over time, developers respond to future expected rules that differ less across states than current rules; failure to measure expected future policy results might would result in coefficients closer to zero than the coefficients would be on permanent rules.

One quick check for timing effects is to remove cities in states that temporarily changed rules in the study period; these are cities in Maryland, Kansas, Illinois, and Ohio. Although only a small number of observations are dropped, they are influential with the "within" estimator. Dropping these observations in the equation in the first column of Table 4 does not markedly change the point estimates, but does render the coefficient on joint and several liability statistically significant at only 10%.

Other covariates. The equations include a number of time-varying covariates in addition to the liability rules. With the fixed effects included in the equation, few of these variables have statistically significant coefficients. Population has a statistically significant positive effect on prices in the column (1) as might be expected, but this effect does not appear in other equations. The other indicators of overall economic conditions — unemployment, manufacturing employment, highway density, and real estate taxes — do not enter with statistically significant coefficients.

The number of lawyers working on contamination for the state enters with a statistically significant positive coefficient in the most equations. The positive coefficient suggests that this variable may capture something other the direct effect of an aggressive program for contaminated sites; an aggressive program would have the same effect as greater private liability, reducing land prices. Additional lawyers may be helpful if they assist developers in attaining rapid approval of cleanup plans and other assurances about the nature of their liability.

3.2 Panel data analysis: Vacancy rates

The second dependent variable of interest is the vacancy rate of industrial space.[12] As above, the estimated equations include city fixed effects and allow an AR(1) process for the errors. Equations are estimated that are limited to center cities and that include suburbs as well.

Liability variables. In the first equation in Table 5 with center cities only, joint and several liability has a statistically significant positive effect on vacancy rates. The magnitude of this coefficient is substantial: it corresponds to about a 40% increase in vacancy rates in the presence of joint and several liability. Although this effect seems large, vacancies may represent a small share of industrial space, so the effect as a share of the full market is less dramatic, accounting for less than 4% of the market. Consistent with the price equations above, the equations do not point to an effect of strict liability on vacancy rates. The point estimate on strict liability is negative, contrary to expectations, and not significant.

In columns (2) and (3), the interaction between joint and several liability and measures of the likelihood of contamination produce positive point estimates, consistent with the idea that joint and several liability is a greater deterrent in places with higher contamination risk. Neither interaction term is individually statistically significant; however, the level and interaction are jointly statistically significant in column (2) with suspected site density, but not in column (3) with historical

[12]SIOR provides both vacant square feet and vacancy rates. I focus on the latter because the data do show dramatic year-to-year changes in available space, presumably due to changes in the definitions employed by the realtors who report each year, whereas vacancy rates exhibit less volatility. In any event, changes in the reporting realtor are unlikely to be systematic. With fixed effects, using the absolute vacant space did not change the conclusions of the analysis.

Table 5: Panel estimates for vacancy rate with fixed effects and AR(1) disturbances

	Dependent variable: Log(Vacancy rate)			
	Center only			All obs
State liability rules				
Strict liability	-.135	-.564	-.218	.239
	(.146)	(.479)	(.331)	(.143)
Strict * Log(Site density)	–	-.108	–	–
		(.125)		
Strict * Log(Old manuf emp)	–	–	.025	–
			(.110)	
Strict * Center city	–	–	–	-.349
				(.196)
Joint and several liability	.353	.801	.282	-.103
	(.147)	(.512)	(.313)	(.148)
Joint and several * Log(Site density)	–	.117	–	–
		(.145)		
Joint and several * Log(Old manuf emp)	–	–	.019	–
			(.104)	
Joint and several * Center city	–	–	–	.461
				(.205)
Other variables				
Log(City population)	-.169	-.206	-.254	-.117
	(.272)	(.297)	(.316)	(.178)
Log(Unemployment rate)	-.041	-.043	-.048	.078
	(.118)	(.122)	(.124)	(.082)
Log(Manufacturing employment)	.110	.168	.182	-.010
	(.263)	(.298)	(.304)	(.184)
Log(Highway density)	-.212	-.211	-.189	.079
	(.180)	(.182)	(.188)	(.101)
Log(Real estate taxes)	.030	.029	.031	-.002
	(.051)	(.052)	(.052)	(.040)
Log(State superfund lawyers)	.050	.053	.053	.109
	(.048)	(.050)	(.051)	(.034)
Log(LCV score)	.159	.164	.160	.126
	(.103)	(.105)	(.106)	(.073)
Log(Abatement cost index)	.059	.100	.111	.136
	(.155)	(.161)	(.162)	(.109)
F-test for strict & strict interaction		.96	.49	1.76
p-value		.38	.61	.17
F-test for J&S & J&S interaction		3.36	2.03	3.40
p-value		.04	.13	.03
ρ for AR(1) process	.60	.59	.59	.56
Number of cities	91	88	87	185
Number of observations	571	556	546	1208

Notes: Standard errors in parentheses.
Not shown: year dummies, dummy for missing highway observations, missing tax, and missing manufacturing employment.

manufacturing employment. As in the price equations, strict liability and its interaction are not jointly statistically significant and the net sign of interactions are inconsistent.

With suburban data added in column (4), joint and several liability and its interaction with center city are jointly significant at 5%. The net effect in center cities continues to be positive as before. Interestingly, the point estimate on joint and several liability outside of central cities is negative, although not statistically significant. A negative effect of joint and several liability on vacancy outside central cities might be consistent with substitution of suburban land in places where urban land is subject to high liability costs.

For strict liability, the coefficient is positive, but only in suburban areas. The effect is not statistically significant, however, so is probably consistent with the general conclusion that strict liability does not have a detectable effect on real estate markets.

Other covariates. As with the price equations, few of the other covariates have statistically significant coefficients in Table 5. One pattern that is interesting is that the variables reflecting state environmental stringency — state superfund lawyers, LCV score, and abatement costs — all increase vacancy rates; these results would be consistent with the somewhat elusive interstate pollution haven effect (Levinson, 1996). However, of these variables, only Superfund lawyers is statistically significant and only in the final equation. This coefficient is consistent with increases in vacant land with more aggressive liability enforcement, but conflicts with the (unexpected) positive effect of this variable in the price equation.

3.3 Endogeneity of liability rules

A nonrandom assignment of liability regimes is a concern for interpretation of the analyses. Although exploiting the panel structure of the data may help to address endogeneity of liability rules, time-varying unobserved heterogeneity remains a potential problem. Liability rules may reflect other unmeasured attributes, such as the amount of public concern about contaminated sites.

The rules may also depend on progress on the brownfields issue if states adjust their rules in ways they hope will encourage redevelopment. However, the choice of liability regime is not mentioned as a factor in brownfields in the policy or legal literature; to my knowledge, the possibility has only been raised in the technical papers of Segerson (1993, 1994). Arguments about the choice of liability regime almost always turn on the trade-off between perceived fairness of expansive liability and the resources it achieves for cleanup. Thus, reverse causality seems to be a less likely source of endogeneity than unobserved heterogeneity.

In this subsection, I explore the endogeneity in the liability rules, using an instrumental variable approach. The previous literature suggests three instruments. First, Alberini and Austin (1999a) study the determinants of liability regimes, focusing on the role of industry mix and environmental preferences. In particular, they find that the number of mining establishments in the state predicts adoption of strict liability, with differential effects for large and small firms. I construct a time series on the number of large and small mining establishments by state from the 1992 and 1997 Census of Mineral Industries, with forward and backward imputation for the remaining years.

Second, Alberini and Austin (2002) find the lagged frequency of accidental spills to affect adoption of liability rules. The idea is that states may react to a flurry of accidents by toughening their liability regimes. Current accidental spills at active facilities should not affect the brownfields problem, which involves past contamination, and thus may be a suitable instrument for this analysis. I construct a variable for the number of spills by state and year from the raw Emergency Response Notification System (ERNS). To mirror Alberini and Austin's measure, I restrict the count of spills to those that occurred at fixed facilities (as opposed to transportation accidents, dumping, and other categories).

Third, Campbell et al. (1998) use total lawyers per capita in a state as an instrument in their analysis of the economic effects of tort reform. The argument for its inclusion is a political one: lawyers have a substantial stake in tort reform and may be major opponents or proponents. Because restriction of joint and several liability was an important component of tort reform over this period, I use this measure. The American Bar Association reports this data at irregular intervals (four times over the period of the data); missing years have been linearly interpolated.

Table 6: Tests of exogeneity of liability rules

	Equation	
	Price	Vacancy rate
Test of exogeneity		
Test statistic	1.96	.40
p-value	.14	.67

Notes: Instruments for liability rules: Lagged spills, lagged mining (small and large) establishments, lagged total lawyers per capita.
Equations as in column (1) in Tables 4 and 5.

When these instruments are used to test for exogeneity of the liability rules, the results fail to reject exogeneity in both equations. Table 6 reports the Davidson-MacKinnon version of the Hausman test for the hypotheses of exogeneity of strict and joint and several liability, using the instruments proposed. The test statistic is moderate for the price equations, leaving the possibility of endogeneity, but very low for the vacancy rate equations.[13]

In evaluating these tests, it is worth noting that the instruments seem relatively successful. Large and small mining establishments have significant first-stage coefficients for both liability regimes. The coefficients on accidental releases are positive and statistically significant for strict liability (as Alberini and Austin (2002) report), but are not statistically significant for joint and several liability. Tests of overidentifying restrictions fail to reject exogeneity of the instruments for the price and vacancy equations, supporting to the validity of the instruments.

[13] If one does run the IV equation on the basis that endogeneity remains a reasonable likelihood for price, the results are disappointing. The coefficient estimates on both joint and several and strict liability are negative as expected, but standard errors are large, especially for joint and several liability.

4 Reported brownfields

The analysis above uses data on the overall industrial real estate market, taking the view that any used industrial land — even that not formally labelled as a brownfield — may be subject to the effects of liability. However, the effects of liability rules on reported brownfields may also be of interest, so this section conducts analyses of these effects.

4.1 Data on reported brownfields

The best available data set on reported brownfield acreages is from surveys conducted by the U.S. Conference of Mayors. Respondents to the USCM survey range from the largest cities to towns of about 10,000 people. The USCM conducted surveys annually between 1997 and 1999 and again in 2002. The total number of reporting cities/towns available for analysis is 366; 25% of the locales are present in three or more years. The survey does not attempt to impose consistency in the definition of brownfields, so the cities' definitions may be quite varied.

The USCM data was matched with the ELI data on the liability regimes. Unfortunately, the narrow time range of the USCM surveys limits the study to cross-sectional identification of the effects of liability rules. During the relevant period, the ELI data on liability rules are available only in 1997 and 2000, with only one transition in liability rules (Arizona eliminated strict liability after 1997). No ELI data are available for 2002, so liability rules are assumed to be the same then as in 2000.

The other covariates are as similar as possible to those used before. Population figures derive from the USCM data itself, so are specific to the reporting locale. For several other characteristics, many locales are too small for city-level data to be available. The USCM locales were therefore matched to one or more counties based on populated place names. Local variables were then assigned based on county-level data, with rates calculated over a multi-county aggregate in the few instances where the populated place spans several counties. These variables include local unemployment rates and the manufacturing share of employment from the BLS.[14] The data on

[14]Manufacturing as a share of employment is used instead of total manufacturing employment because the employ-

Table 7: Summary statistics for USCM data set, by joint and several liability

	All obs		Without J&S		With J&S	
	Median		**Median**		**Median**	
Brownfield acres	100		100		115.5	
	Mean	**St dev**	**Mean**	**St dev**	**Mean**	**St dev**
Brownfield acres	723	3964	375	727	829	4509
Joint and several liability	.765	.424	0	–	1	–
Strict liability	.853	.354	.787	–	.873	–
Metropolitan population (thousand)	195	626	165	197	204	707
Unemployment rate (%)	4.78	1.91	5.26	2.78	4.63	1.52
Manufacturing share of employment	.152	.071	.134	.060	.157	.073
Taxes forgone (2002 $/acre)	44.0	133	30.1	75.6	47.8	144
Contaminated sites/ sq mile	.236	.449	.216	.624	.243	.380
Historical manuf employ / sq mile	185	356	78.3	184	217	389
State superfund lawyers per million	.894	.947	.660	.794	.964	.978
League of Conservation Voters score	52.3	21.2	43.2	15.1	55.1	22.0

density of suspected contaminated sites and historical manufacturing data discussed earlier was also merged by county.

A measure of local real estate taxes was constructed from the USCM data. Respondents to the survey provide a range for the estimated tax loss from the failure to redevelop brownfields. Dividing the midpoint of this range by the acres of brownfields provides a measure of the tax rate for the locale. This tax rate may measure not only real estate taxes, but also anticipated sales and wage taxes if the property were developed in the way the city would like.

State characteristics used in the earlier equations are also included. The equations include the average LCV score for the state's House delegation and the per capita number of contaminated site lawyers working for the state (from ELI).[15]

Table 7 provides summary statistics for the full data set and for the subsets with and without joint and several liability. In the full data set, the cities claim an average of 723 acres of brownfield sites. The average city has a population of 195,000, but the median is lower because the range in city size goes up to 8 million (New York).

ment data are at a county-level and thus may not conform well to the size of the locale reporting the brownfields.

[15]Because ELI data are not available for 2002, the lawyers data for this year is assigned from 2000. The pollution abatement cost index used previously as a measure of environmental stringency would have to be entirely extrapolated for this data set, so is not used.

A large difference appears in reported mean brownfield acres between the cities with and without joint and several liability. Although the mean brownfield acres in the joint and several cities is much larger, the distributions of acres appear almost identical until the 95th percentile, where the joint and several cities include a few cities reporting tens of thousands of acres. Both groups have medians (reported in the first row of Table 7) of about 100 brownfield acres.[16]

Cities with joint and several liability differ from the other cities along a number of dimensions. The former are larger, more industrial, and have more suspected contaminated sites.[17] Unlike in the earlier data, joint and several liability is also associated with more aggressive contaminated site programs, as measured by the number of state superfund lawyers. These cities are also located in greener states, as represented by the average LCV score for the state.

4.2 Results with reported brownfields data

Table 8 reports the results of panel data analyses of the USCM survey. In the equations, only the years 1997 through 1999 are used because they are within the range of the ELI data. In the final column, data for 2002 are added, assuming that rules are the same as in 2000. A number of cities joined the panel in 2002, so adding the extra year's data expands the geographic coverage. Because only one liability rule changed, identification of these coefficients comes almost entirely from the cross-section and only random effects are included. The equations allow within-panel AR(1) errors as before.

Liability variables. In the first column in Table 8, the coefficients on the liability rules show a similar to pattern to the pattern found in the overall vacancy rate. Joint and several liability has a statistically significant and surprisingly large effect in raising the number of acres of brownfields. The coefficient of .510 corresponds to 67% more brownfields with joint and several liability. The

[16] Dropping cities reporting more than 10,000 acres did not substantively change the estimates in the next subsection.

[17] The average density of contaminated sites is much greater in this data set (.24 per square mile) than in the general real estate market data (.09 per square mile). The disparity is largely in the upper tail; the medians are similar (.07 versus .05 respectively). The difference seems to result from greater ability to pinpoint counties in the USCM data set. For example, the highest values in the USCM data (3 sites per square mile) are for cities located in a single county in Northern New Jersey. In the earlier data, a handful of Northern New Jersey counties are in a single observation.

Table 8: Panel estimates for brownfield acreage with city random effects and AR(1) disturbances

	Dependent variable: Log(Brownfield acres)		
	1997–99	**1997–1999, 2002**	
State liability rules			
Strict liability	-.057	-.148	-.227
	(.269)	(.233)	(.237)
Joint and several liability	.510	.554	.417
	(.237)	(.207)	(.203)
Other variables			
Log(City population)	.763	.761	.741
	(.098)	(.086)	(.081)
Log(Unemployment rate)	.139	.154	.271
	(.240)	(.220)	(.222)
Log(Manuf share of employment)	.030	.068	.115
	(.224)	(.197)	(.192)
Log(Tax rate)	-.325	-.307	–
	(.038)	(.031)	
Log(Superfund site density)	-.064	-.088	-.165
	(.139)	(.124)	(.121)
Log(Historical manuf employment)	.154	.200	.252
	(.178)	(.162)	(.158)
Log(State superfund lawyers)	.194	.073	.095
	(.108)	(.086)	(.084)
Log(LCV score)	-.154	-.187	-.312
	(.205)	(.140)	(.148)
1998	.193	.151	-.060
	(.129)	(.116)	(.116)
1999	.277	.237	.085
	(.141)	(.134)	(.135)
2002	–	.446	.179
		(.170)	(.161)
Constant	-1.46	-.996	-.215
	(1.90)	(1.70)	(1.62)
ρ for AR(1) process	.27	.68	.57
Number of cities	257	305	366
Number of observations	386	521	663

Notes: Standard errors in parentheses.
Not shown: dummy for missing lawyer data.
In final column, 2002 liability rules and state Superfund lawyers assigned 2000 values.

point estimates thus suggests a larger effect than the 40% increase found for vacancy rates; the comparison may be consistent with stronger liability effects on sites with greater likelihood of contamination. On the other hand, the coefficient on strict liability is not statistically significant and has a very small point estimate.

The next two columns of the table include the 2002 survey, expanding the data set, but relying on extrapolated liability rules. With the inclusion of 2002, the coefficient on joint and several liability is statistically significant at the 5% level and again large in magnitude.

A concern with this analysis is the role of the tax rate variable, which enters with a counter-intuitive negative, but very precisely estimated, coefficient. To construct this variable, reported foregone taxes are divided by the number of brownfield acres to calculate a tax rate. However, the consequence is that the inverse of the left-hand-side variable is on the right-hand-side. The final equation in the table drops the tax variable to avoid this problem. The point estimate falls somewhat with this exclusion, but remains statistically significant. About half of the reduction in the point estimate results from including observations previously excluded for lack of tax data.

The equations in Table 8 are weaker evidence of an effect of liability rules than the earlier equations because it is not possible to use fixed effects to control for heterogeneity and because cities may have very different definitions of brownfields. However, the consistency with the early results (showing a deterrent effect of joint and several liability but not of strict liability) suggests robustness for these results.

Other variables. The relationships of reported brownfields acreage with some of the other variables are also interesting. Reported brownfield acreage increases with the city's population, but with an elasticity less than one. This coefficient suggests that the smaller cities face greater relative burdens from brownfields than larger cities, all else equal. The regressions do not point to any relationship with unemployment rates or the manufacturing share of employment.

Somewhat surprisingly, the number of sites reported to the Superfund inventory also does not have a statistically significant coefficient and its point estimate is negative. The number of inventory sites has sometimes been used as measure of the number of brownfields (e.g., Simons, 1998). This result suggests that it does not agree well with city governments' assessment of their brownfields problem. Old manufacturing employment fairs somewhat better as a predictor of reported brownfields, with a positive coefficient. However, the coefficient is still not statistically significant and far below the unitary elasticity one might expect.

Finally, the coefficients on the two state environmental stringency variables have signs that suggest differing effects. On the one hand, more state superfund lawyers per capita raises the number of brownfields, perhaps because more aggressive programs identify more sites or raise the costs of developing contaminated sites. On the other hand, states with higher LCV scores have fewer brownfields acres (in all but the first equation). The latter effect could be the result of more stringent controls on the behaviors that give rise to contamination or of more extensive previous cleanups.

5 Conclusions

The results of the empirical analysis are consistent with the view that joint and several liability not only drives down industrial real estate prices, but also increases the vacancy of industrial land. Both the price effect and quantity effects are concentrated in central cities, as might be expected. One cannot rule out the possibility of substitution of greenfields for brownfields in cities with joint and several liability, but the estimated equations do not provide positive evidence of this effect. In addition, the results provide little support for either a price or a quantity effect from strict liability: I speculate that standards for due care are sufficiently high or uncertain that negligence rules provide little protection from liability. In analysis of a limited data set on reported brownfields, joint and several liability is associated with more brownfields, but strict liability is not.

The results thus suggest that liability is at least partially capitalized but still deters redevelopment. The reason for the deterrence may be a general problem, such as adverse selection or the possibility that parties are judgement proof. It may also be specific to joint and several liability.

With either type of cause, the results provide an argument for reducing reliance on joint and several liability. However, joint and several liability does have advantages that should be weighed against these costs. It provides the government with greater resources for cleanup and may facilitate settlement (Chang and Sigman, 2002). A targeted approach that provides protection from joint and several liability only when properties are sold might therefore be more desirable than broader liability relief.

References

[1] Alberini, Anna and David Austin. On and Off the Liability Bandwagon: Explaining State Adoptions of Strict Liability in Hazardous Waste Programs, *Journal of Regulatory Economics* 15 (1999), 41–63.

[2] Alberini, Anna and David H. Austin. Strict Liability as a Deterrent in Toxic Waste Management: Empirical Evidence from Accident and Spill Data, *Journal of Environmental Economics and Management* 38 (1999), 20–48.

[3] Alberini, Anna and David Austin. Accidents Waiting to Happen: Liability Policy and Toxic Pollution Releases, *Review of Economics and Statistics* 84 (2002), 729–41.

[4] Alberini, Anna, Alberto Longo, Stefania Tonin, Francesco Trombetta, and Margherita Turvani. The Role of Liability, Regulation and Economic Incentives in Brownfield Remediation: Evidence from Surveys of Developers, *Regional Science and Urban Economics,* 35 (2005), 327–51.

[5] American Tort Reform Association. http://www.atra.org/states/ (Last viewed: August 16, 2005).

[6] Boyd, James, Winston Harrington and Molly K. Macauley. The Effects of Environmental Liability on Industrial Real Estate Development, *Journal of Real Estate Economics and Finance* 12 (1996), 37–58.

[7] Boyle, Sue, New Jersey Department of Environmental Protection, personal communication, June 28, 2005.

[8] Campbell, Thomas J., Daniel P. Kessler, and George B. Shepherd. The Link Between Liability Reforms and Productivity: Some Empirical Evidence, *Brookings Papers on Economic Activity: Microeconomics* (1998), 107–137.

[9] Chang, Howard F. and Hilary Sigman. Incentives to Settle Under Joint and Several Liability: An Empirical Analysis of Superfund Litigation, *Journal of Legal Studies* 29 (2000), 205–36.

[10] Chang, Howard F. and Hilary Sigman. The Effect of Joint and Several Liability Under Superfund on Brownfields. NBER Working Paper 11667, 2005.

[11] Cooter, Robert and Thomas Ulen. *Law and Economics.* Glenview, IL: Scott, Foresman and Company, 1988.

[12] Good, David and Kenneth Richards. Looking Inside the Black Box: Microlevel Empirical Analyses of the Impact of State and Federal Policy Instruments on Hazardous Waste Generation and Management. 2002 Progress Report. http://es.epa.gov/ncer/index.html (last viewed: May 18, 2004).

[13] Grimski, Detlef and Uwe Ferber. Urban Brownfields in Europe. *Land Contamination and Reclamation* 9 (2001), 143-148.

[14] Jackson, Thomas O. Environmental Contamination and Industrial Real Estate Prices, *Journal of Real Estate Economics Research* 23 (2002), 180–99.

[15] Levinson, Arik. Environmental Regulations and Manufacturers' Location Choices: Evidence from the Census of Manufactures, *Journal of Public Economics* 62 (1996), 5–29.

[16] Levinson, Arik. An Industry-Adjusted Index of State Environmental Compliance Costs in *Behavioral and Distributional Effects of Environmental Policy* (Carlo Carraro and Gilbert E. Metcalf, editors), 2000.

[17] McGrath, Daniel T. Urban Industrial Land Redevelopment and Contamination Risk, *Journal of Urban Economics* 47 (2002), 414–42.

[18] Nijkamp, Peter, Caroline A. Rodenberg, and Alfred J. Wagtendonk. Success Factors for Sustainable Urban Brownfield Development, A Comparative Case Study Approach to Polluted Sites, *Ecological Economics* 40 (2002), 235–252.

[19] Noonan, Frank and Charles A. Vidich. Decision Analysis for Utilizing Hazardous Waste Site Assessments in Real Estate Acquistion, *Risk Analysis* 12 (1992), 245–251.

[20] Pendergrass, John A. *An Analysis of State Superfund Programs: 50-State Study, 2000 Update,* Washington, DC: Environmental Law Institute, 2001.

[21] Probst, Katherine N., Don Fullerton, Robert E. Litan, and Paul R. Portney. *Footing the Bill for Superfund Cleanups: Who Pays and How?* Washington, DC: Resources for the Future, 1995.

[22] Reisch, Mark. *Superfund and Brownfields in the 107th Congress.* Washington, DC: Congressional Research Service, 2003.

[23] Robertson, Heidi Gorovitz, One Piece of the Puzzle: Why State Brownfields Programs Can't Lure Business to the Urban Cores without Finding the Missing Pieces, *Rutgers Law Review* 51 (1999), 1075–1132.

[24] Schoenbaum, Miriam. Environmental Contamination, Brownfields Policy, and Economic Redevelopment in an Industrial Area of Baltimore, Maryland. *Land Economics* 78 (2002), 60–71.

[25] Segerson, Kathleen. Liability Transfers: An Economic Assessment of Buyer and Lender Liability, *Journal of Environmental Economics and Management* 25 (1993), S64–S65.

[26] Segerson, Kathleen. Property Transfers and Environmental Pollution: Incentive Effects of Alternative Policies, *Land Economics* 70 (1994), 261–72.

[27] Simons, Robert A. *Turning Brownfields into Greenbacks: Developing and Financing Environmentally Contaminated Real Estate* Washington DC: Urban Land Institute, 1998.

[28] Stafford, Sarah L. Assessing the Effectiveness of State Regulation and Enforcement of Hazardous Waste, *Journal of Regulatory Economics* 23 (2003), 27–41.

[29] Tietenberg, Tom H. Indivisible Toxic Torts: The Economics and Joint and Several Liability, *Land Economics* 65 (1989), 305–19.

[30] US Conference of Mayors. *Recycling America's Land: A National Report on Brownfields Redevelopment, Volume IV* USCM, 2002.

[31] Urban Institute, et al. *The Effects of Environmental Hazards and Regulation on Urban Redevelopment.* Washington, DC: The Urban Institute, 1997.

[32] Wernstedt, Kris, Lauren Heberle, Anna Alberini and Peter Meyer. The Brownfields Phenomenon: Much Ado about Something or the Timing of the Shrewd? Resources for the Future Discussion Paper, 2004.

[33] Wernstedt, Kris, Peter B. Meyer, and Anna Alberini. Attracting Private Investment to Contaminated Properties: The Value of Public Interventions, *Journal of Policy Analysis and Management* 25 (2006), 347–369.

[34] Wheaton, William C. and Raymond G. Torto. An Investment Model of Demand and Supply of Industrial Real Estate, *AREUEA Journal* 18 (1990), 530–548.

[35] Wooldridge, Jeffrey M. *Econometric Analysis of Cross-Section and Panel Data* Cambridge, MA: MIT Press, 2002.

[36] Zabel, Jeffery. The Economics of Brownfields — with an Application to Somerville, Massachusetts, Working Paper, Tufts University, 2003.

The Risk-Based Approach to Brownfield Redevelopment:

Is Less Cleanup Better?

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Abstract: *This paper contains a model of the market for brownfields, properties with actual or perceived contamination, when redevelopment is encouraged by a Risk-Based Approach (RBA). Such an approach allows some contamination to remain on site if fewer people will be exposed to the contamination, such as when the intended reuse is non-residential. We derive land market efficiency conditions if complete cleanup is efficient as well as if only partial cleanup is efficient. For each case, we consider the efficiency and distributive effects of two types of RBA: buyer-only eligible and buyer and seller eligible. When the market internalizes full cleanup, while social efficiency calls for only partial cleanup, buyer-seller RBA achieves efficiency. However, relieving the seller of some liability for cleanup goes against the polluter-pays principle of equity. Buyer-only RBA, the predominant form of RBA, is not efficient, encouraging too many sales. (JEL Topic Area Codes: Q 28, Q 24, K32)*

The Risk-Based Approach to Brownfield Redevelopment:

Is Less Cleanup Better?

I. Introduction

The "Small Business Liability Relief and Brownfields Revitalization Act" signed into law January 11, 2002, defines a brownfield site as "real property, the expansion, redevelopment, or reuse of which may be complicated by the presence or potential presence of a hazardous substance, pollutant, or contaminant."[1] The 2002 act is indicative of a change in federal policy initiated in 1995, when the USEPA announced its original Brownfields Action Agenda in response to the widespread perception that CERCLA (Comprehensive Environmental Response, Compensation, and Liability Act of 1980, better known as "Superfund") liability had deterred redevelopment.

In the absence of redevelopment, occupants of surrounding areas would continue to suffer the impact of any pollutants, such as through soil or groundwater contamination. The direct justification for brownfield redevelopment is pollution reduction. There are also widely cited secondary justifications, among which are to slow the development of greenfields, land not previously developed beyond that of agriculture or forestry use, and to revitalize inner cities where many brownfields are located. Related benefits include the slowing of sprawl, an expanded tax base, and the creation of jobs.

The USEPA has encouraged the states to develop their own approaches. A majority of the states have adopted a Risk-Based Approach (RBA), a "flexible" approach that allows some contamination to be left in place for non-residential development, on the rationale that exposure to contaminants will be less than for residential reuse. Many, but not all, of the state programs, are modeled on Risk-Based Corrective Action (RBCA or "Rebecca"), which develop standards and look-up tables of required corrective action based on health impacts to surrounding areas of designated brownfield redevelopment land uses, or on Tiered Approach to Correction Action (TACO), where one of three tiers is chosen for the standard, depending upon the impact on human health. Given that risk-based programs are Voluntary Cleanup Programs (VCPs), developers can choose to apply for the standard, but are not required to do so. While environmentalists and community activists express concern that some contaminants remain, proponents of a flexible approach maintain that requiring a pristine standard for all uses results in the abandonment of marginal properties for which cleanup costs exceed the value of the redeveloped property, which results in higher pollution.

To support the use of risk-based programs, USEPA and the individual states cite "success stories" showing the transformation of former brownfields into desirable reuses. However, there has been little systematic analysis. The stories focus on buyer/developer outcomes, but neglect the seller/original owner role. One consequence is that, while some programs may extend the incentives to the original owner, the implicit preference is aimed at redevelopers who were not responsible for the contamination. Hence, the programs may encourage property transactions that may or may not be efficient.

In a 2004 conference on "Estimating Community Economic Impacts from the Reuse of Contaminated Properties," sponsored by Resources for the Future, Industrial Economics, Inc. and the National Center for Environmental Economics, Wernstedt (2004) examines fifteen studies and finds them to have a wide range of values and typically an absence of an economic framework.[2] In commenting on papers, Kerry Smith (2004) notes the absence of a basic demand/supply model. He proposes that hedonic studies can help with demand, while developer's profits can shed light on supply. In a separate report, Vitulli, Dougherty, and Bosworth (2004) find that the EPA and state programs do not currently collect the data that would be needed to allow analysis of the factors that bear on potential redevelopment.

This paper provides a model of the real-estate market to compare the effects on real-estate price, quantity, efficiency, and equity, of risk-based standards restricted to buyer/developers as compared to programs where the owner/polluter is also eligible. Throughout the paper, we assume that states set the partial cleanup standard based on an evaluation of the marginal costs and marginal benefits of incomplete cleanup, marginal cost in the form of health repercussions from remaining contaminants, and marginal benefits from increased redevelopment and partial cleanup, as opposed to no cleanup. Nevertheless, future work should explicitly address the determination of optimal cleanup, especially in light of the possibility that such a standard will differ for different properties and different locales.

As in papers by Boyd, Harrington, and MacAuley (1996), Segerson (1997), Corona and Segerson (2005), and Schwarz and Hanning (2005), the purpose of the model in this paper is to analyze possible inefficiencies in the real-estate market for contaminated properties brought about by CERCLA. Those papers consider a variety of potential land-market inefficiencies, such as uncertainty, imperfect detection of contamination, judgment-proof buyers, and asymmetric information on the amount of contamination between buyer and seller. The focus in this paper is on the purported justification of RBA aimed at optimal cleanup, and that complete cleanup for non-residential properties is inefficient.[3]

Some commentators have expressed concerns about risk-based programs. For example, Hamilton and Viscusi (1999) base their flexible approach to Superfund liability on the impact of a site on surrounding properties, rather than aiming relief at how the site itself is redeveloped. A "flexible" approach could also leave the door open to greater political influence on setting the non-residential standard.[4] Additionally there are long-term monitoring issues to ensure that requirements are met, and that future land uses are consistent with the level of cleanup.[5]

But perhaps most fundamentally, risk-based standards do not directly address the primary arguments for why CERCLA is thought to unduly discourage brownfield reuse, such as uncertainty (unpredictability of magnitude of cleanup, and who is liable) and asymmetric information (such as the possibility that the seller knows more about the contamination level than does the buyer).[6] It also leaves unsettled the extent to which the buyer who qualifies for reduced cleanup from RBA also accepts liability, including a higher expected liability in such forms as reopeners by the state and civil suits from third parties if only a partial cleanup takes place. Simons, et al (2003), find reopeners to be extremely rare to date, although they caution that they may increase over time. Wernstedt et al. (2006) find that the value to developers of relief from third-party liability is 40% higher than the value of relief from current cleanup. Such factors need to be recognized in order to predict the effects of risk-based policies, and in order to make policy recommendations as to the efficiency of these policies, and to features of the policies, such as whether or not the original owner should be eligible. However, it is first necessary to focus on the purported justification for RBA, that developers should be allowed to leave some contamination in place for non-residential redevelopment. This initial question needs to be resolved before considering the use of RBA when there are additional sources of potential inefficiency which RBA was not designed to correct.

In Part II, we develop a model of the real-estate market appropriate to incorporating risk-based incentives. We develop a market for brownfields where buyers inherit a portion of liability, but where aside from liability issues, there are no other potential sources of market failure. The purpose is to consider the extent to which the real-estate market will achieve private and social efficiency. Private efficiency indicates that the property is redeveloped by whoever – buyer or seller—values the property more, and that the efficient number of properties are redeveloped, while social efficiency indicates that there is the right amount of cleanup on each property. We then introduce RBA, first with only buyers eligible for reduced liability, and then where buyers and sellers are eligible, to examine possible inefficiencies of reduced cleanup in a market that was initially efficient. We also consider equity consequences in the form of income changes for brownfield buyers and sellers.

In Part III, we reexamine the real-estate market when social efficiency calls for partial, rather than complete, cleanup. Once we identify any private or social inefficiencies, we again consider buyer-only and buyer-seller eligibility for reduced cleanup. It is of policy interest to see if a risk-based approach leads to more sales when it is efficient to have a sale. Risk-based incentives may go too far, encouraging sales even when a sale is not efficient. Or they may not go far enough, increasing sales in an efficient direction but stopping short of the efficient point. Or RBA may increase sales, but reduce cleanup per sale. It also is possible that RBA may not affect efficiency, but primarily transfer income by reducing cleanup costs. Part IV concludes.

II. **Modeling the Effects of RBA When the Complete Cleanup is Efficient**

a. Privately and Socially Efficient Land Market

Proponents of brownfield redevelopment justify the policy as improving social efficiency by reducing exposure to contaminants, encouraging an increase in non-residential development to increase the use of inner-city property, and reducing the use of greenfields. It is less often justified as a correction for inefficiency in the private real-estate market. Nevertheless, there can be a concern that the real estate market for contaminated land might lead to less than the efficient amount of redevelopment. Advocates of RBA as a stimulus to redevelopment focus on the potential for inefficiency of brownfield redevelopers who are required to fully clean up the site when less than complete cleanup may be optimal, such as for non-residential reuses.

We begin by presenting a private, competitive real estate market for brownfields, and consider whether or not it is privately and socially efficient. It is privately efficient if sales only take place when the value of the land to the buyer is at least as great as the value to the seller. It is socially efficient if the optimal number of sites are cleaned up, and cleanup on those sites is at the optimal level. If the market fully internalizes the externality, government intervention might not be necessary.[7] We then introduce RBA--buyer-only and buyer-seller eligible-- and consider its effects on private and social efficiency as well as distribution of income among buyers, sellers, and government.

A sale is privately efficient so long as:

$V_b \geq V_s$ (1)

where V_b and V_s are the values placed on the brownfield by a buyer/ potential developer and by a seller/ current owner, gross of contamination liability L.[8]

An existing owner will sell the property iff:

$$P_1 - (1-\alpha)L \geq V_s - L \qquad (2)$$

where P_1 is brownfield market price, L is liability for cleanup, V_s is owner's valuation of the property, gross of cleanup liability, V_s –L is minimum willingness-to-accept (WTA), and α is the buyer's share of liability (and (1- α) is owner's remaining share if the property is sold).[9]

Solving for price P_1:

$$P_1 \geq V_s - \alpha L \qquad (3)$$

The buyer condition is:

$$V_b - \alpha L \geq P_1 \qquad (4)$$

where V_b is buyer's value gross of contamination, and

$V_b - \alpha L$ is maximum willingness to pay (WTP).

For a sale to take place,

$$V_b - \alpha L \geq P_1 \geq V_s - \alpha L \qquad (5)$$

Assuming a competitive market with perfect information where both the buyer and the seller know α and L with certainty, a sale will take place so long as $V_b \geq V_s$. Therefore, in a competitive market including perfect information, the land market achieves private efficiency despite the presence of contamination in the market.

Social efficiency requires not only that the right number of properties Q^* are cleaned up, but that each is cleaned to an efficient level, which is initially L^*. Total cleanup is L^*Q^*.

If RBA is introduced on the premise that the brownfield real-estate market is inefficient due to contamination, the premise is in error if the real-estate market for brownfields is privately and socially efficient. The market will perfectly internalize the cleanup cost, with market price decreasing as the buyer's share of liability increases. If the buyer absorbs all liability, the market price will fall by the full amount of the liability.

As an example, consider a property worth $30,000 to a buyer and $25,000 to the seller. The contamination liability cost is $10,000, and the buyer will be responsible for 40% of the cleanup. In this case, market price will be $21,000 if price equals seller minimum WTA. Then, the seller would be indifferent between continuing to own the property, worth $15,000 net of liability, or selling the property and contributing $6000 towards liability. The buyer pays $25,000 in all, $21,000 for the property and $4000 towards liability. Only efficient sales will occur; a buyer who valued the property at less than $25,000 would not pay $21,000 for the property. Total cleanup is $10,000 for each property that is redeveloped.[10]

We now introduce RBA to consider its effects on such a market.[11]

b. RBA in a Privately and Socially Efficient Land Market

i. Buyer-Only Eligible

RBA is generally envisioned as giving liability relief to a buyer who is not currently the owner of the property. We consider this version first, followed by the effects of RBA if it is available to current owners as well as potential buyers.

The condition for a seller who is not eligible for RBA is:

$$P_2 - (1-\alpha)cL \geq V_s - L \qquad (6)$$

where c is the buyer's fraction of cleanup required.[12] Solving for price P_2:

$$P_2 \geq V_s - \alpha L(c + \frac{1-c}{\alpha}) \qquad (7)$$

For $\alpha < 1$, $c + (1-c)/\alpha > 1$ and so $P_2 < P_1$. Seller WTA decreases with the introduction of RBA, even though the seller is ineligible for reduced liability. By selling the property, the owner potentially reduces liability by more than when there is no RBA, if the developer use is non-residential.

The buyer condition is:

$$V_b - \alpha cL \geq P_2 \qquad (8)$$

Without RBA, the buyer condition was:

$$V_b - \alpha L \geq P_1$$

Since $\alpha cL < \alpha L$, $P_2 > P_1$. Developer WTP increases.

For a sale to take place,

$$V_b - \alpha cL \geq P_2 \geq V_s - \alpha L(c + \frac{1-c}{\alpha}) \qquad (9)$$

The rhs < lhs by $(1-c)L$. It is now possible for $V_b < V_s$, and yet a sale will take place when $V_b \geq V_s - L(1-c)$. Such a sale is privately inefficient when $V_b < V_s$.

Total cleanup is cLQ_2, as compared to optimal cleanup L^*Q^*. Since $Q_2 > Q^*$ and $cL < L^*$, for $c < 1$, total cleanup with buyer-only RBA could increase or decrease, as compared to the market without RBA. Even if the total cleanups are equal, there is no assurance that buyer-only RBA produces efficient cleanup, since it produces less than efficient cleanup on a greater than efficient number of properties.

As an example, let the value to the developer fall to \$24,000, and suppose the developer eligible for RBA is required for cleaning up 80% of the contamination, a decrease from 100% in the absence of RBA. All other values are the same as in the earlier numerical example. Owner minimum WTA is now $\$25{,}000 - .4(10{,}000)[.80 + (.20/.4)] = \$19{,}800$. The owner is willing to accept less than without RBA, since the owner gains from a sale to a non-residential developer by reducing the remaining liability after the sale. The developer pays \$23,000 in all, \$19,800 towards the purchase price and \$3200 in liability costs. Even though value to the developer is less than to the owner, a sale takes place. Cleanup per property is \$8000, but the number of cleaned up properties would be larger.

As compared to no RBA, developer maximum WTP increases, while owner WTA decreases. So the probability that a trade will take place increases. But increased transactions should not be taken as a proxy for increased efficiency. As indicated, some of the transactions will be inefficient, where the land was more valuable to the seller than to the buyer.

Buyer-only RBA leads to the possibility of privately inefficient sales. It creates an incentive for sellers as well as buyers for non-residential redevelopment. Finally, it redistributes income to buyers and sellers.

ii. Buyer and Seller Eligible for RBA

The condition for selling the property is:

$$P_3 - (1-\alpha)cL \geq V_s - cL \quad (10)$$

which differs from RBA restricted to the buyer in that the seller who redevelops the land need only clean a percentage 'c' of the liability.

Solving for price P_3:

$$P_3 \geq V_s - \alpha cL \quad (11)$$

Compare $V_s - \alpha cL$ to $V_s - \alpha L$, the original condition without RBA. For $c < 1$, lhs > rhs. So $P_3 > P_1$. Since we found earlier for the seller that $P_2 < P_1$, $P_3 > P_1 > P_2$. So making both potential buyer and current owner eligible for RBA increases the seller's minimum WTA above pre-RBA market price, while if only buyers are eligible for RBA, price was shown to be below the pre-RBA market price. As compared to a privately efficient brownfield market with no RBA, RBA (buyer only eligible) is likely to lower market price while RBA (buyer and seller eligible) is likely to increase price. Extending the benefit of RBA to the seller increases the opportunity cost of selling the land, resulting in a higher reservation price.

The buyer condition is:

$$V_b - \alpha cL \geq P_3 \quad (12)$$

Compared to P_1 (no RBA):

$V_b - \alpha cL > V_b - \alpha L$, so $P_3 > P_1$.

Comparing P_3 and P_2 (equations (12) and (9)), we have the same buyer condition, so the buyer's maximum willingness to pay is unaffected by including the current owner as eligible for RBA. Since the owner's minimum willingness to accept is higher when the owner is eligible for RBA, and the buyer's maximum willingness to pay is unaffected, transactions are less likely as compared to when only buyers are eligible.

In order for a sale to occur:

$$V_b - \alpha cL \geq P_2 \geq V_s - \alpha cL \qquad (13)$$

With both buyer and seller eligible for RBA, any sales that take place will in fact be privately efficient, since it is necessary that $V_b \geq V_s$ for a sale to occur. Both owner WTA and developer WTP increase equally, resulting in an efficient outcome as was the case in the model without RBA.

Price is higher than in the absence of RBA, but the number of sales will be identical. Clearly, however, the seller is better off, as compared to no RBA, getting a higher price and the potential for a lower liability payment. Those who favor "polluter pays" may not like this redistribution of income. Nevertheless, unlike the version of RBA that restricts benefits to the potential developer, there is no private inefficiency.

From a social efficiency perspective, total cleanup is $cLQ^* < LQ^*$. While buyer-seller RBA leaves the number of sales unchanged as compared to the private market, it results in a less than socially optimal level of cleanup, in the case where full cleanup is efficient.

If we consider the previous numerical example, seller minimum WTA is now \$21,800, while buyer maximum WTP is unchanged. A potential buyer who values the property at \$24,000 would no longer buy a property that now costs

$25,000. Only developers who value the property at $25,000 or more would buy the land, so only efficient transactions will take place. Cleanup per property remains at $8000, as with buyer-only RBA, but fewer properties are cleaned up than with buyer-only RBA.

III. Modeling the Effects of RBA When Partial Cleanup is Efficient

a. Socially Inefficient Land Market

The premise for RBA is that optimal cleanup should reflect marginal benefit, as reflected by population exposure to remaining contaminants. The premise is that fewer people will be exposed if the land is redeveloped non-residentially (e.g. commercially or industrially, with industrial redevelopment associated with the smallest exposure). In the absence of a policy such as RBA, the land market will reflect complete cleanup. However, buyer-only and buyer-seller RBA affect the market differently. It remains to be seen as to which one is preferable to correct a market failure due to buyers and sellers basing price on complete cleanup, rather than optimal cleanup. Assuming the two forms have differing effects, it may still be that even the more efficient form does not fully coincide with the efficient condition.

If the market transaction reflected optimal cleanup, the seller condition becomes:

$$P_4 - (1-\alpha)cL \geq V_s - cL \qquad (14)$$

Solving for P_4:

$$P_4 \geq V_s - \alpha cL \qquad (15)$$

Since $\alpha cL < \alpha L$ for $c < 1$, $P_4 > P_1$.

The equations are identical to equations (10) and (11), where buyer-seller RBA was imposed in a privately and socially efficient land market. Seller minimum willingness to accept increases, as compared to when buyer and seller are required to do a complete cleanup.

The implication is that buyer-seller eligible RBA corrects the land market social inefficiency of "too much" cleanup. Buyer-seller RBA results in total cleanup of cLQ^*, which is less than private cleanup LQ^*. Both produce a privately efficient outcome, but only buyer-seller RBA produces a socially efficient outcome.

It is worth emphasizing that with buyer and seller sharing liability-related cleanup costs, allowing less than complete cleanup raises the seller's price. Price increases by $\alpha L - \alpha cL = (1-c)\alpha L$, so that as 'c' decreases, price increases. As more contamination is allowed to remain on site if the seller redevelops, selling price increases.

The buyer condition is:

$$V_b - \alpha cL \geq P_4 \qquad (16)$$

which is identical to the buyer-seller RBA condition imposed on a privately and socially efficient land market. Buyer price is higher than without RBA, and increases by exactly the same amount as the increase in the seller's price.

The equilibrium condition is:

$$V_b - \alpha cL \geq P_4 \geq V_s - \alpha cL \qquad (17)$$

Sales only take place when $V_b \geq V_s$, the private market efficiency condition. Cleanup per property is cL rather than L. This outcome suggests that buyer-seller eligible RBA will correct a market failure due to full (too much cleanup) rather than partial cleanup.

ii. Buyer-Only Eligible

The equilibrium condition obtained earlier in equation (9) was:

$$V_b - \alpha cL \geq P_2 \geq V_s - \alpha L(c + \frac{1-c}{\alpha}) \qquad (9)$$

and the rhs < lhs by (1-c)L, so that a sale takes place when

$V_b \geq V_s - L(1-c)$.

As was discussed earlier, buyer-only RBA leads to the result that sales will occur when $V_b < V_s$, which is privately inefficient. Buyer-only RBA encourages more sales, but doesn't reflect marginal cleanup benefit cL. Instead, the condition contains (1-c)L. For 'c' = 1, the sale condition returns to $V_b \geq V_s$. But as 'c' decreases, allowing increasing contamination to stay in place, the number of inefficient sales increases. Buyer-Seller RBA was efficient. Buyer-only RBA, which is the predominant form of RBA, is not efficient.[13]

It may well turn out that if there are other sources of market failure, such as uncertainty or asymmetric information, judgment proof defendants, or imperfect detection of brownfield contamination. It would then be necessary to evaluate both types of RBA in the presence of different market failures, as well as multiple market failures. But since the primary intent of RBA is to correct a supposed market failure due to requiring too much cleanup, it is likely that there exists a better tool than RBA for correcting other market failures.

IV. Summary and Conclusions

Public policy has become more favorably inclined towards the redevelopment of brownfields, with the primary purpose of reducing pollution, as well as to reduce greenfield use and rejuvenate inner-city areas. States have developed a variety of programs intended to lessen the obstacles to redevelopment. The risk-based approach (RBA), for example, allows non-residential redevelopment to meet a less stringent standard than the more conservative residential standard. The purpose of this paper was to examine the effects of brownfield incentives in the form of reduced liability when considering their effects on both the buyer and the seller of brownfields.

In particular, our results suggest that efficiency increases if buyers as well as sellers are eligible for RBA, in contrast to most programs that restrict the incentive to buyers. Buyer-only eligibility results in a higher price than when there is no RBA. Output is unaffected, but cleanup is at the optimal level, which is less than full cleanup for non-residential properties.While such a policy may go against the equity of "polluter pays," it leaves more redevelopment in the hands of the original owner, who has better information about the property, and leads to fewer transactions and therefore lower transactions costs. Buyer-seller RBA reduces the price of the property as compared to no RBA, and increases the number of sales. Some sales occur where the seller values the property more than the buyer, which is privately inefficient. It would be more efficient for these properties to be developed by the current owner, rather than to sell the property.

RBA has the stated purpose of encouraging greater development. It offers a more flexible approach, with a more lenient standard for non-residential property. In addition, it can establish who is liable and for how much. The incentives are generally aimed at buyers, rather than the original owner of the property who was responsible for the contamination. Given that the effects of RBA have received little attention, it is worth examining a number of variants, such as whether only buyers are eligible, or both buyers and sellers are eligible. A second variant is whether the liability is shared, or whether the buyer who accepts a less stringent standard is now fully liable.

Incorporating a real estate market sheds light on the effects of brownfield policy reforms. First, it is necessary to distinguish whether the reforms are meant to correct a private inefficiency in the brownfields real-estate market, or whether the justification is social inefficiency, which could be due to non-optimal cleanup, as well as concerns about over-development greenfields and unused or underused inner city properties. Second, the effects of the brownfield policies will depend upon the absence or presence of market failure in the brownfield real-estate market. Third, the effects will depend upon the form of the incentives. In the case of RBA, are both buyer and seller eligible? Does RBA shift the liability to the buyer?

Federal, state, and local governments have been devoting considerable resources to the redevelopment of brownfields. It is time for policymakers to devote some of these resources to considering whether or not the money is being well spent.

References

Boyd, J., W. Harrington, and M. MacAuley.1996. "The Effects of Environmental Liability on Industrial Real Estate Development," *Journal of Real Estate Finance and Economics*, 12, 37-58.

Breggin, L.1999. *A Guidebook for Brownfield Property Owners*, Environmental Law Institute.

Coase, R.1960. "The Problem of Social Cost," *Journal of Law and Economics*, 3 (4), 1-44.

Corona, Joel and Kathleen Segerson. (2005). "Brownfield Redevelopment under the Threat of Bankruptcy," Presentation at Southern Economics Association, Washington, DC. October.

Geisinger, A.2001. "Rethinking Risk-Based Environmental Cleanup," *Indiana Law Journal*, 76 (2), 367-402.

Hamilton, J. and K. Viscusi.1999. "How Costly is "Clean"? An Analysis of the Benefits and Costs of Superfund Site Remediations," *Journal of Policy Analysis and Management*, 18 (1), 2-27.

Haner, N. C.2004. "Cashing in on brownfields: Expanding border will benefit some, but others see red over "blight"." *Orlando Business Journal* (May).

Kennedy, E., W. Smathers, Jr. and R. Costa.2002. "Market Incentives for Protecting Red-Cockaded Woodpecker Habitat," *Issues in Wildlife Economics and the Ecology-Economics Interface*, Southern Natural Resource Association Proceedings, 153-162.

Meyer, P., 2000. "Approaches to Brownfield Regeneration: The Relative Value of Financial Incentives, Relaxed Mitigation Standards and Regulatory Certainty," paper prepared for Track 8, Urban Development and Management, Association of European Schools of Planning Congress, Brno, Czech Republic.

Probst, K. and K. Wernstedt.2004. "Estimating Community Economic Impacts from the Reuse of Contaminated Properties: Workshop Summary and Highlights," Workshop organized by RFF in conjunction with Industrial Economics, Inc. (IEc) and NCEE, (Feb).

Schwarz, P. and A. Hanning (2005). "The Risk-Based Approach to Brownfield Redevelopment: Model and Simulation," paper presented at Southern Economic Association, Washington, DC.

Segerson, K. 1997. "Legal Liability as an Environmental Tool: Some Implications for Land Markets," *Journal of Real Estate Finance and Economics*, 15 (2), 143-159.

Simons, R. A., J. Pendergass and K. Winson-Geideman. 2003. "Quantifying Long-Term Environmental Regulatory risk for Brownfields: Are Reopeners Really an Issue? *Journal of Environmental Management*, 46 (2), 257-269.

Smith, V. Kerry. 2004. "Measuring Property Values: Comment," Conference on Estimating Community Economic Impacts from the Reuse of Contaminated Properties, Resources for the Future, http://www.rff.org/rff/Events/Estimating-Community-Economic-Impacts-from-the-Reuse-of-Contaminated-Properties.cfm

Wernstedt, K. 2004. "Overview of Existing Studies on Community Impacts of Land Reuse," Conference on Estimating Community Economic Impacts from the Reuse of Contaminated Properties, Resources for the Future, http://www.rff.org/rff/Events/Estimating-Community-Economic-Impacts-from-the-Reuse-of-Contaminated-Properties.cfm

Wernstedt, Kris, Peter B. Meyer, Anna Alberini, and Lauren Heberle. 2006. “Incentives for Private Residential Brownfields Development in US Urban Areas.” *Journal of Environmental Planning and Management*, 49(1). 101-119.

[1] See Public Law 107-118 (H.R. 2869).

[2] The conference materials are at http://www.rff.org/rff/Events/Estimating-Community-Economic-Impacts-from-the-Reuse-of-Contaminated-Properties.cfm . Probst and Wernstedt (2004), in a summary of the workshop, conclude "It is clear that all government officials—at the federal, state, and local levels—face pressure to demonstrate positive impacts from their programs. And yet, discussion among the participants evince disagreement on how to frame these impacts in terms of national and/or social welfare. Most participants would agree that consistent and accurate measurement of reuse impacts is useful for gauging program success and for targeting future efforts, but there is not yet a clear constituency for this approach or someone within EPA currently tasked with coordinating the measurement of site reuse impacts across Agency programs."

[3] Meyer (2000) fully describes four categories of direct interventions: regulatory relief (where he includes RBA and institutional controls such as deed restrictions, liability reduction (lowering the potential for suits, and increased availability of liability insurance), direct financial support (loans, grants, assistance for transactions costs such as site assessment), and site reclamation by the state. A fifth intervention he sites that is often overlooked is constraining greenfield development, of which Portland, OR is representative.

[4] Noelle Haner in the May 14, 2004, Orlando Business Journal, writes about an attempt to declare most of downtown Orlando a brownfield, so as to be eligible for financial incentives to redevelop. In any case, surveys show individuals place a much higher priority on the cleanup of hazardous wastes than do experts, so there is considerable latitude in setting commercial and industrial standards. See Gayer and Viscusi (2002), who examine the effect of reported news on hazardous waste on housing prices.

[5] State regulators are concerned that property subject to institutional controls (such as deed restrictions or zoning) which has previously been cleaned up to industrial standards has been converted to inappropriate (i.e., residential) use; see John S. Applegate and Stephen Dycus (1998). Geisinger (2001) points out the difficulty of accurately predicting future uses, and suggests the need for parties to post bond and some form of insurance in case the best future use requires a higher clean-up standard. Tight state budgets also limit long-term monitoring.

[6] While the approach appears to be more flexible than requiring a single standard, an approach that allows for some form of trading might increase efficiency still further. A trading approach has been applied to such land–related issues as wetlands and habitat for endangered species. See, for example, USEPA National Forum on Water Quality Trading, July 22 and 23, 2003 and Kennedy, Smathers, and Costa (2002).

[7] Essentially, the insight from Coase (1960) that has come to be known as the Coase Theorem is that in the absence of transactions costs, negotiations between parties can internalize externalities and lead to an efficient outcome. All that is needed is a clear definition of property rights, which may require minimal government intervention in the form of courts defining property rights. Given that transactions costs do exist, the so-called Coase Corollary looks to the courts to give the rights to the highest-cost avoider of the externality. The lower-cost avoider will then negotiate to the efficient outcome, if it is different from the one established by the courts.

[8] One can think of these values as the land value of a property where there is neither perceived nor actual contamination, such as a greenfield that is identical to a brownfield, except there is neither perceived nor actual contamination.

[9] Note that if the land is not sold, the current owner is fully liable.

[10] The maximum possible market price is $26,000, were price equal to the developer's maximum wtp.

[11] In a separate paper, Schwarz and Hanning (2005) consider RBA when there are private inefficiencies in the real-estate market such as asymmetric information on contamination.

[12] Note that the current owner receives a share of the reduced liability if the owner sells the property, but is fully liable if the property is not sold.

[13] In Segerson (1996), shared liability drove the inefficient outcomes, in the presence of a market failure due to asymmetric information. But here, shared liability α is not part of the equilibrium condition, and so is not driving the inefficiency. The inefficiency is attributable to extending a benefit to only one side of the market.

DRAFT

Brownfield Redevelopment Under the Threat of Bankruptcy

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Abstract

This paper evaluates firm behavior in the context of brownfield redevelopment. A simple land development choice model can help determine how firms make brownfield and greenfield choices, as well as assess how government policy affects firm actions. Some brownfield investment occurs despite the transfer of liability. The threat of bankruptcy can lead to an adverse selection problem, where developers facing lower inherent profitability choose to develop brownfields over greenfields. In general, the market outcome can yield more, less, or the same amount of development as the efficient level. A subsidy can induce efficient brownfield investment when the brownfield development externality is sufficiently large relative to the developer's potential liability and wealth. If the externality is relatively small, a second instrument such as a greenfield tax is needed to achieve the first-best outcome.

Journal of Economic Literature Classification:

Keywords: brownfields, environmental liability, hazardous waste site

I. Introduction

A brownfield site is "real property, the expansion, redevelopment or reuse of which may be complicated by the presence or potential presence of a hazardous substance, pollutant or contaminant."[1] Common wisdom holds that redeveloping brownfields preserves pristine greenfields, lessens urban sprawl, while additionally relieving urban blight. As undeveloped land has become increasingly scarce within urban areas, both the public and private sectors have turned to brownfield redevelopment as a solution.

While firms often experience private benefits such as lower transportation costs, closer access to labor markets, and reduced land prices when locating new projects on brownfield properties, they do not realize the full social benefits resulting from the positive externalities associated with redevelopment. This leads to an inefficient market. In addition, firms are wary of brownfields because of the uncertainty surrounding the liability costs attached to these properties, and the potential such costs have for bankrupting firms.[2] These disincentives for private brownfield development create a role for government support to develop brownfields.

Government intervention may target market inefficiencies in different areas. Inefficiencies may be found in the land market for brownfields, the lending market for redevelopment projects, or in the decision-making process of the developers themselves. Under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) of 1980, current brownfield owners can be held liable for past environmental contamination, regardless of their contribution to contamination. This can lead to distortions in the land market, as future owners would not wish to inherit such environmental liability.[3] Banks have expressed

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[1] See http://www.epa.gov/brownfields/ for more information or http://www.epa.gov/brownfields/sblrbra.htm on the Small Business Liability Relief and Brownfields Revitalization Act, source of the original definition.

[2] The U.S. Environmental Protection Agency's (EPA) Brownfield website provides information regarding liability issues. See http://www.epa.gov/brownfields/liab.htm .

[3] Boyd, et al. (1996) and Segerson (1993, 1994, 1997) examine inefficiencies in the land market stemming from brownfield environmental liability concerns. While this literature attributes most market inefficiencies to asymmetric information regarding environmental contamination, Segerson (1997) does note that land markets can be inefficient even in the context of symmetric information because of the judgment-proof problem. Alberini et al. (2005) and Wernstedt et al. (2006) empirically find that developers value liability relief. See also Chang and Sigman (2005) and Schwarz and Hanning (2005) for recent theoretical work on liability and the land market.

concern over lender liability for redevelopment projects financed through their loans.[4] In addition to affecting the real estate and lending markets, liability can have an influence on the developer's decision-making process. This area has not received as much attention in the economic literature.

In this paper, we develop a land development choice model in which there is perfect capitalization of liability into land prices. We use the model to ask whether in this context there is a role for government policy to promote brownfield redevelopment. More specifically, we ask whether perfect capitalization will lead to brownfield redevelopment even in the absence of any government policy. Our goal is to see whether the justification for government policies rests solely on imperfect capitalization or whether instead there is some other basis for brownfield policies.

Our land development decision model results in firms making their development decision according to the level of project-profitability they face. In certain cases, a range of firms facing medium-profitability projects choose brownfield redevelopment even in the absence of any government policy. Furthermore, these same firms facing medium-return projects make this choice while risking the potential for bankruptcy, while solvent firms facing high-return projects choose instead to favor greenfield investment. This adverse selection problem occurs because bankruptcy relieves the medium range of firms from bearing the full risk of environmental liability.

The market outcome can yield more, less, or the same brownfield investment as the efficient level, depending on the extent of the external benefit from brownfield development, as well as the factors determining the developer's potential liability, and the developer's wealth. In addition, if banks can not observe developer type and are lead by brownfield bankruptcy concerns to charge a higher rate for development projects on brownfields than on greenfields, then the market leads to inefficient greenfield investment.

In the latter part of the paper, we examine the impact of a generic government subsidy policy designed to promote brownfield redevelopment, and ask whether a subsidy can be used to induce efficient development decisions. We focus our attention on subsidy programs because of their simplicity and popularity. We show that, while a subsidy will increase brownfield

[4] Boyer and Laffont (1997), Feess and Hege (2000), Heyes (1996), and Pitchford (1995) are examples of the lender liability literature. See Balkenborg (2001), Lewis and Sappington (2001), and Pitchford (2001) for a reaction to

development, it will not necessarily induce efficient development decisions by itself. We show that the subsidy expands both the upper and the lower end of the range of firms choosing brownfield redevelopment. When the external benefit of brownfield redevelopment is sufficiently large relative to liability conditions and wealth, a subsidy can alone induce efficient brownfield and greenfield investment. However, if the brownfield development externality is not sufficiently large, then an additional instrument such as a greenfield tax is needed to achieve efficient investment decisions.

II. Socially Efficient Development

Consider a developer who is considering investing in a development project that will yield a gross return of *a*, where $0 \leq a \leq A$. This return depends on characteristics of the project and/or developer that are not publicly known. Thus, while the developer knows *a*, it is not known by others.

The project can be undertaken on a contaminated brownfield (denoted by the index *B*), on a pristine Greenfield (denoted by the index *G*), or not at all. While there is some (known) expectation regarding the extent of the contamination, the exact extent cannot be determined until cleanup is underway. Hence, at the time of the development decision, the exact level of contamination (and hence required cleanup) of the brownfield property is unknown. We assume that with probability $z>0$ the brownfield has a high level of contamination, requiring a high level of cleanup (L_H), and with probability ($1-z$) it has a low level of contamination, requiring a low level of cleanup (L_L). Expected cleanup is thus given by $\bar{L} = zL_H + (1-z)L_L$.

Let V_o denote the value of the land to the current owner, where this value is measured gross of any liability for cleanup. The social return from the project, ignoring any external costs or benefits from developing on a greenfield vs. a brownfield, is simply $a - r$, where *r* is the social cost of the resources (other than land) needed for the project. In addition, we assume that development of a greenfield generates external costs of $S_G \geq 0$ as a result of urban sprawl, while the development of a brownfield generates an external benefit

Pitchford (1995).

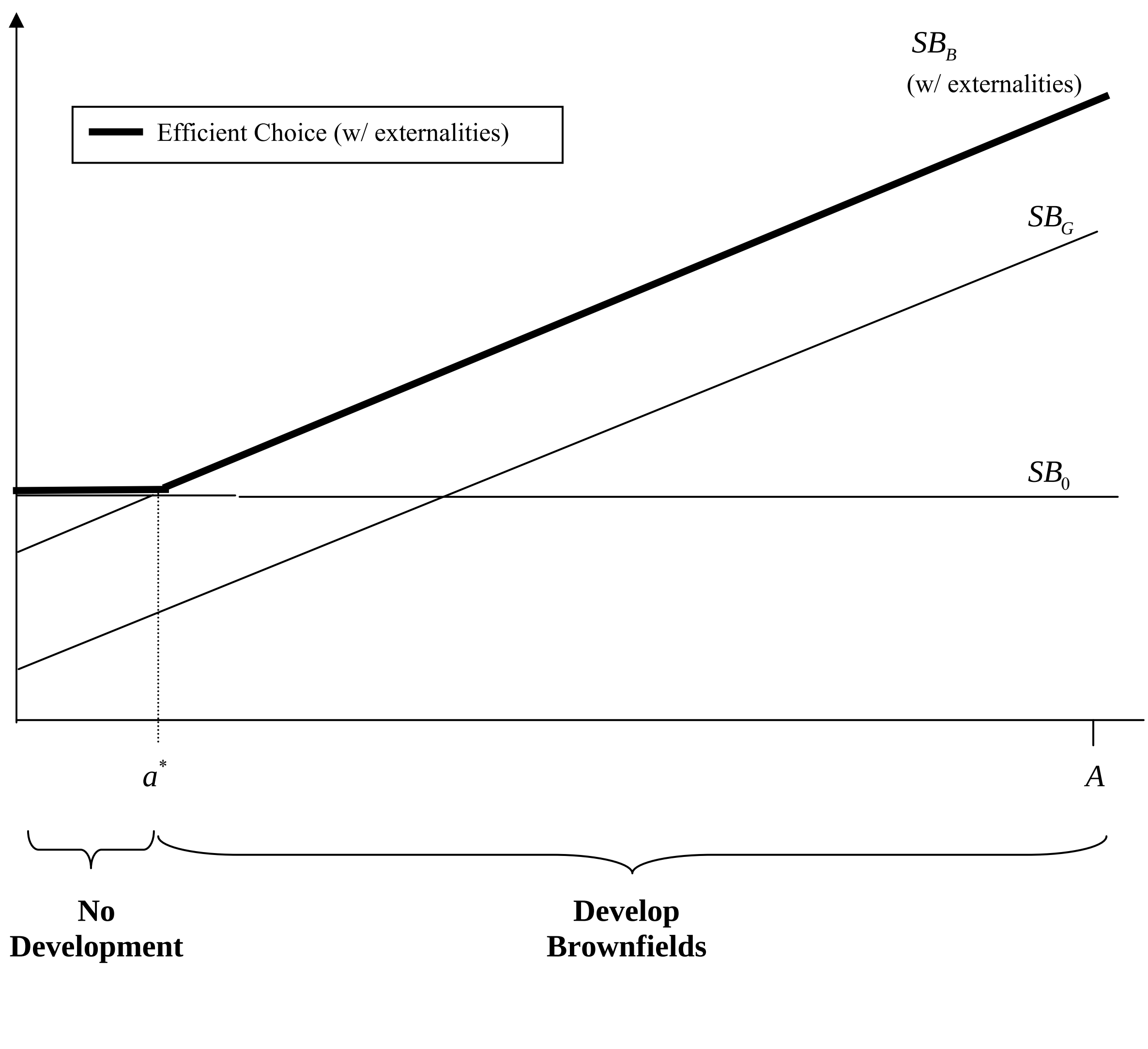
$S_B \geq 0$ as a result of urban renewal. Given this, the expected net social return of developing on the brownfield, given by $a - r - \bar{L} + S_B$, always exceeds the expected net social return of developing on a greenfield, given by $a - r - \bar{L} - S_G$. Note that, even if the development proceeds on the greenfield, the expected damages from contamination $\bar{L}$ are still borne by society (since contamination of the brownfield is not eliminated by developing the greenfield) and hence still enter the calculation of expected net social returns. Thus, under our assumptions it is never efficient to develop a greenfield.[5] It would, however, be efficient to develop the brownfield if and only if the expected social return from doing so is at least as high as the expected social return if no development occurs, given by $V_o - \bar{L}$. Hence, it is efficient to develop the brownfield if and only if

$$a \geq V_o + r - S_B . \tag{1}$$

This defines a threshold level of a, namely, $a^* \equiv V_o + r - S_B$, above which brownfield development is efficient. The socially efficient development decisions are illustrated in Figure 1.

[5] While this is perhaps an extreme case, it serves to capture the essence of a situation where society would like to encourage brownfield development over greenfield development.

Figure 1- Social Efficiency



III. Market Incentives for Development

We turn next to the representation of market incentives for investment in brownfields vs. greenfields. Clearly, absent any policy intervention, any external costs or benefits in the form of sprawl or urban renewal are not reflected in market incentives. Our primary interest, however, is in another potential distortion caused by the possibility that high liability might bankrupt a developer of a brownfield.

Let $V_D^i(a)$ denote the value of the land to the developer, given his type and the land type (*i=B,G)*, measured gross of the price paid for the land and any associated liability for cleanup. We assume that, if the property is sold to the developer, a fraction $0 < \alpha \leq 1$ of the responsibility or liability for cleanup is transferred to the new owner (developer), while the remainder of the cleanup remains the responsibility of the current owner (seller). Greenfield properties have no associated contamination and hence no cleanup or liability-related costs.

Let P_B be the price of a brownfield and let P_G the price of a greenfield. Then a necessary condition for the owner of a brownfield to be willing to sell and the developer to be willing to buy is:

(2) $$V_D^B(a) - \alpha \bar{L} \geq P_B \geq V_o - \alpha \bar{L} .$$

Likewise, a necessary condition for the owner of a greenfield to be willing to sell and the developer to be willing to buy is:

(3) $$V_D^G(a) \geq P_G \geq V_o .$$

We assume that there are many possible greenfields and brownfields available, and as a result, the prices are determined by the minimum prices the owners would accept. This implies that

(4) $$P_B = V_o - \alpha \bar{L},\ P_G = V_o \text{ and } P_B = P_G - \alpha \bar{L} .$$

Note that this implies that the transfer of expected liability is perfectly capitalized into the price of the brownfield, i.e., the price is reduced (relative to the greenfield price) by the expected liability that the developer assumes.

The developer is assumed to finance the inputs necessary for the project through a loan from a bank. While the bank cannot observe the developer's type, it has the ability to determine if a loan is for a brownfield or greenfield project, and charges interest rates r_B and r_G for each. The developer uses as collateral its initial wealth *w*. We assume that bankruptcy occurs if contamination is high and the developer's asset level is low. We capture the potential for high liability (but not low liability) to bankrupt the developer by assuming that

$$\text{(5)} \qquad \alpha L_H > w - r_B - P_B > \alpha L_L .$$

This implies that even a developer with the lowest type (*a=0*) has sufficient collateral that he will not be bankrupted by low liability. However, developers without sufficient assets (from initial wealth or returns from the project) would be bankrupted by high liability. This occurs if *a* is sufficiently low, given the other parameter values, i.e., if

$$\text{(6)} \qquad a < r_B + P_B + \alpha L_H - w = r_B + V_o + \alpha(1-z)(L_H - L_L) - w \equiv \hat{a} .$$

The assumption in (5) assumes that $\hat{a} > 0$. In addition, provided $r_B \geq r_G$, it implies that, regardless of type, the developer will not be bankrupted by development of a greenfield.[6] We make this assumption to focus on bankruptcies triggered by high liability coupled with low project returns rather than bankruptcies triggered by low project returns alone.

Note that, from the developer's perspective, the difference between the brownfield and the greenfield stems solely from the transfer of liability and its potential to bankrupt the developer. If $\alpha = 0$, i.e., if no liability is transferred, then there is no potential for bankruptcy (i.e., (5) would never be satisfied) and hence we would expect the bank to charge identical interest rates for both types of projects, i.e., $r_B = r_G$. In this case, the developer's returns on the

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[6] Theory would suggest brownfield interest rates would exceed greenfield interest rates because of the additional perceived or actual risk involved with brownfields. Anecdotal evidence supports this claim (see Haughney (2006)).

brownfield and greenfield are identical, and he is indifferent between the two. However, when $\alpha > 0$, the potential for high liability to bankrupt some types of developers suggests that the bank would charge a higher interest rate for loans to brownfield developers, with the "premium" dependent on the extent of the liability transfer.[7] Thus, we assume $r_B = r_B(\alpha, z)$ with $\partial r_B / \partial \alpha > 0$ and $\partial r_B / \partial z > 0$.

We can now identify the expected returns from the three options open to the developer: (i) forego development, (ii) develop a greenfield, and (iii) develop a brownfield. Clearly, the return from no development is simply $\pi_o = w$, while the return from greenfield development is given by:

(7) $$\pi_G = w + a - r_G - P_G .$$

Development of a brownfield yields the following expected return:

(8) $$E\pi_B = \begin{cases} w + a - r_B - P_B - \alpha \bar{L} & \text{if} \quad a \geq \hat{a} \\ (1-z)(w + a - r_B - P_B - \alpha L_L) & \text{if} \quad a < \hat{a} \end{cases} .$$

The developer chooses the option that yields the highest (expected) return. This varies with the developer's type. Clearly, whenever $\alpha > 0$ (so that $r_B > r_G$), $\pi_G > E\pi_B$ if $a \geq \hat{a}$. Thus, developers with high profitability projects prefer to develop greenfields. However, when $a < \hat{a}$, then $\pi_G > E\pi_B$ if and only if $w + a - r_G - P_G > (1-z)(w + a - r_B - P_B - \alpha L_L)$, i.e., if and only if

(9) $$a > \hat{a} - \frac{1}{z}(r_B - r_G) \equiv a_B^G ,$$

where $a_B^G < \hat{a}$. This implies that a developer with $a < a_B^G$ gets a higher expected return from developing a brownfield than from developing a greenfield. However, at sufficiently low levels

[7] This could occur because a greater liability transfer will expand the range of types that face potential bankruptcy. Of course, if the developer's bankruptcy in turn transfers liability to the lender, the bank could have an additional

of a, even development of the brownfield becomes unprofitable (i.e., yields a lower expected return than foregoing development altogether). This occurs when

(10) $$a < \hat{a} - \alpha(L_H - L_L) + \frac{w}{1-z} \equiv a_B^o.$$

Note that $a_B^0 \leq a_B^G$ if the difference between high and low contamination levels is sufficiently large, i.e., if

(11) $$\alpha(L_H - L_L) \geq \frac{w}{1-z} + \frac{1}{z}(r_B - r_G).$$

We focus the remainder of our discussion on this case in order to show how the transfer of liability and the potential for bankruptcy that results can affect development decisions.

We summarize the optimal decision for the developer in the following proposition.

Proposition 1: *The developer's optimal development strategy is as follows:*

(12) $$\begin{cases} \text{develop a greenfield} & \text{if } a \geq a_B^G \\ \text{develop a brownfield} & \text{if } a_B^o \leq a < a_B^G \\ \text{forego development} & \text{if } a < a_B^o \end{cases}.$$

Note that the thresholds in Proposition 1, and hence the development decisions, will depend on the associated liability, in particular the magnitudes of z, α, and $L_H - L_L$.

The payoffs from the three options and the optimal development strategy are illustrated graphically in Figure 2. Note that the optimal strategy induces a sorting of developers by their type, where high types develop greenfields, middle range types develop brownfields, and low types do not develop at all. The following results follow immediately.

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incentive to increase interest rates to brownfield developers who assume some liability for cleanup. For a discussion of this latter possibility and its effect on interest rates, see Heyes (1996).

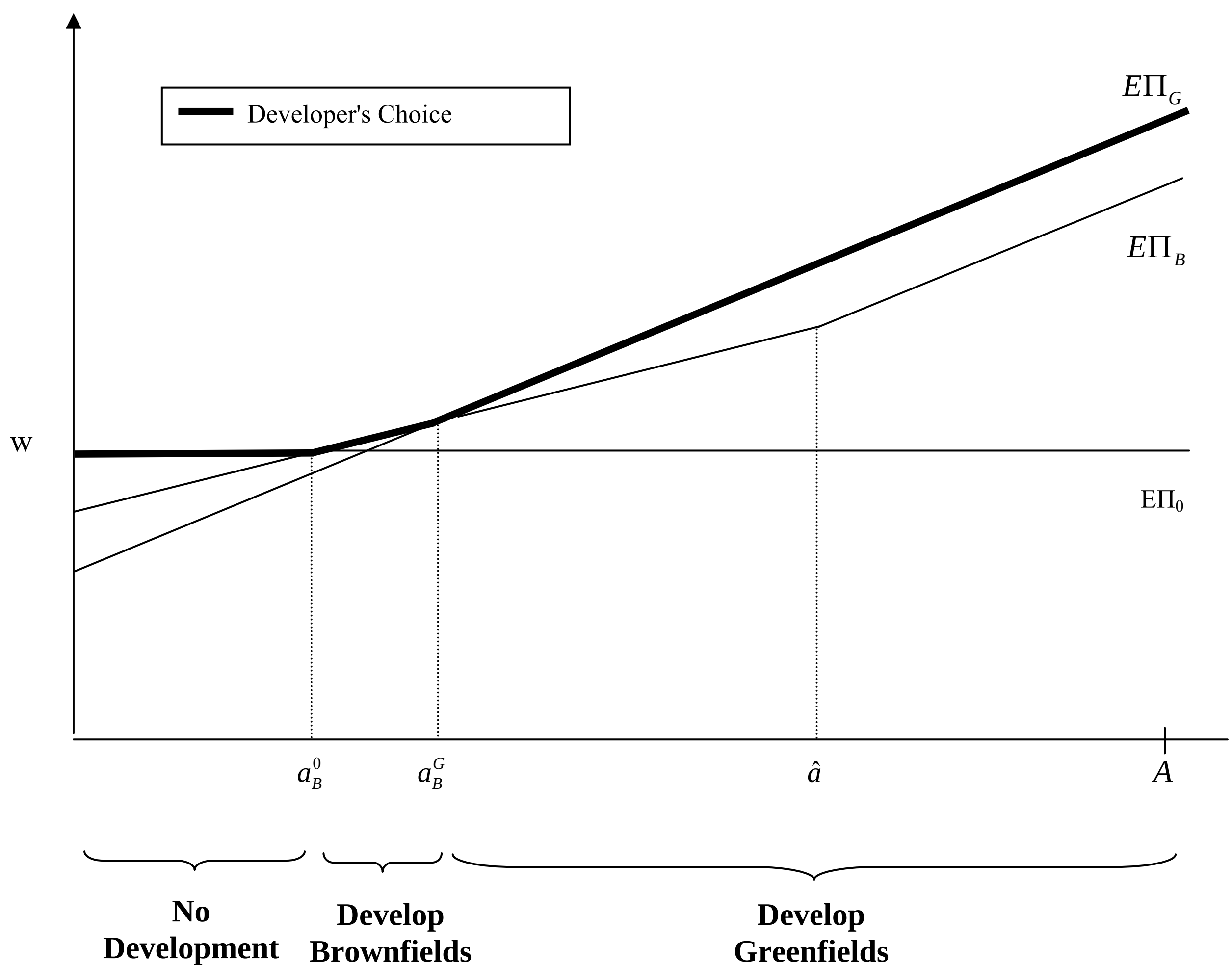
Corollary 1: *Even in the absence of any government incentive, some brownfield redevelopment occurs, despite the transfer of liability.*

Corollary 2: *Developers who choose brownfield development have projects of lower inherent profitability (i.e., lower a) than developers who choose greenfield development.*

Corollary 3: *All developers who choose brownfield development face bankruptcy if contamination is high.*

Corollary 4: *All developers who do not face the possibility of bankruptcy, i.e., who have sufficient assets to avoid bankruptcy even when contamination is high, choose greenfield development.*

Figure 2- Firm Development Decision



These results imply an adverse selection problem. Under perfect capitalization of liability costs into the land market, regardless of their type, all buyers of brownfield properties receive a discounted price on the land that reflects the expected cleanup costs that are transferred to them (and hence no longer borne by the current owner) upon purchase/sale of the property. This discount is in turn the amount that a high-ability developer would expect to pay in liability or cleanup costs, since he is always solvent and hence always pays for the cleanup costs. However, because low-ability developers are bankrupted if liability is high, even though they receive a discounted price that reflects the transferred share of expected liability, they only pay actual cleanup costs when those costs are low. Thus, the discount they receive in the brownfield price exceeds the cleanup costs they expect to pay, since they realize that they will pay only if those costs are low. This makes the transferred liability less costly to a lower-ability developer than to a high-ability developer. When this benefit is sufficient to offset the higher interest rate paid on the brownfield as well as the risk of losing both his collateral and the project's profits (a loss that decreases with *a*), it will induce brownfield development by the lower-ability developers.

IV. The Role of Brownfield Development Subsidies

Having characterized development decisions in the presence of potential bankruptcy, we turn next to a comparison of the market and efficient outcomes to see whether subsidy-type policies that decrease the costs of brownfield development can in principle induce efficient decisions.[8]

Consider first the decision to develop at all. It is clear from the above analysis that this decision will be efficient if and only if $a^* = a_B^0$. In generally, however, this condition will not hold, given $z>0$ *and* $S_B>0$, *unless by chance* $S_B = \alpha z(L_H - L_L) - \frac{z}{1-z} w - (r_B - r)$. There are two distortions that cause the market outcomes to deviate (generally) from the socially efficient decisions. The first is the presence of the external benefit S_B from development of brownfields (e.g., urban renewal), which is not internalized by the developer. This causes private incentives for development (of brownfields) to be too low, which tends to lead to too little development. On the other hand, the bankruptcy potential implies that firms that are subject to this threat will receive a discount in the price of a brownfield that actually exceeds their expected liability (since the price discount is based on expected cleanup costs that the current owner would bear while the developer actually pays the associated liability only when it is low and hence not sufficient to bankrupt the developer). This effect tends to lead to too much development, given (11). The combination of these two effects determines whether overall market investment is greater or less than the efficient level (or possibly equal to it if the two effects are exactly offsetting).

Proposition 2: *In general, the market outcome can yield more development, less development, or exactly the same amount of development as the efficient level, depending on the extent of the external benefit from brownfield development, the factors determining the extent of the developer's potential liability, and the developer's wealth.*

The above proposition identifies a market inefficiency at the "low end" of the distribution of a. In addition, there is obviously a market inefficiency at the "high end" of the distribution of a. Even though greenfield development is by assumption inefficient (since brownfield development generates positive spillovers through urban renewal while greenfield development generates negative spillovers through urban sprawl), by Proposition 1, developers with sufficiently high values of a, i.e., $a \geq a_B^G$, develop greenfields,. This results from the adverse selection problem and the associated higher interest rate that even high-a developers have to pay for brownfield development. If the interest rate were the same on both property types, then $\hat{a} = a_B^G$ (see (9)). In this case all developers who face bankruptcy if liability is high would develop brownfields (if they develop at all), while all developers who would not be bankrupted by high liability are indifferent between brownfield and greenfield development.

[8] We choose to focus on a subsidy for simplicity, because of its appeal to developers (Alberini et al. (2005) and Wernstedt et al. (2006)), and because of the myriad of federal and state level grant programs (Bartsch and Dorfman (2000), Bartsch and Wells (2005)).

Proposition 3: *If banks are unable to observe developer type and charge a higher rate for development projects on brownfields than on greenfields because of the potential for bankruptcy, then the market outcome leads to inefficient (too much) greenfield development.*

Having established the inefficiency of market development at both the high and the low end of the distribution of *a* when the potential for liability-related bankruptcy exists, we now turn to the question of whether a subsidy for brownfield development could correct these market inefficiencies. We consider a generic subsidy designed to increase the profitability of projects undertaken on brownfields. One possibility, of course, is simply to provide a subsidy sufficiently large to eliminate the potential for bankruptcy if liability is high. While this would clearly eliminate the greenfield development incentive for high-*a* developers (as well as encouraging development at the low end), such a subsidy is likely to be very costly, with the magnitude of this cost depending not only on the magnitude of the required subsidy but also on the social cost of funds. For this reason, we are interested in considering smaller subsidies that do not eliminate the potential for bankruptcy, but might nonetheless encourage brownfield development. If we let *s* be the (fixed) subsidy paid to a developer who develops a brownfield, this implies that $s < \alpha L_H + r_B + P_B - w$. If we redefine $\hat{a}$ to reflect the receipt of the subsidy, i.e., let

(5’) $$\hat{a} \equiv r_B + P_B + \alpha L_H - w - s,$$

then assuming *s* is not sufficiently large to eliminate bankruptcy ensures $\hat{a} > 0$. Clearly, since $\partial \hat{a} / \partial s < 0$, an increase in the subsidy reduces the range of developers who face potential bankruptcy.

In examining the impact of such a subsidy on brownfield development, it is clear that the subsidy will affect the returns from brownfield development. Expected returns can now be written as

(8’) $$E\pi_B = \begin{cases} w + a + s - r_B - P_B - \alpha \overline{L} & \text{if} \quad a \geq \hat{a} \\ (1-z)(w + a + s - r_B - P_B - \alpha L_L) & \text{if} \quad a < \hat{a} \end{cases}$$

where $\hat{a}$ is now defined by (5'). Note that *s* has a larger impact on returns for developers who do not face potential bankruptcy than it has on developers who would be bankrupted by high liability. The reason, of course, is that the developer effectively loses the subsidy if he is bankrupted, and hence when $a < \hat{a}$ the subsidy contributes to expected returns only with probability *(1-z)*.

Graphically, the effect of the subsidy is to shift the curve for $E\pi_B$ in Figure 2 upward. It is clear that this will have three effects. First, it will create a windfall for developers who would have chosen brownfield development even in the absence of a subsidy. Second, it will induce some (but not necessarily all) developers who chose greenfield development before to now switch to brownfield development. With the subsidy, the threshold for greenfield development becomes

(9') $$a > \hat{a} - \frac{1}{z}(r_B - r_G - s) \equiv a_B^G,$$

where

$$\frac{\partial a_B^G}{\partial s} = \frac{\partial \hat{a}}{\partial s} + \frac{1}{z} = \frac{1-z}{z} > 0 .$$

Thus, as expected, the subsidy discourages greenfield development.

The third effect of the subsidy is to induce some (but not necessarily all) developers who had previously chosen to forego development altogether to now choose brownfield development. With the subsidy, the threshold for no development becomes

(10') $$a < \hat{a} - \alpha(L_H - L_L) + \frac{w}{1-z} \equiv a_B^o$$

where

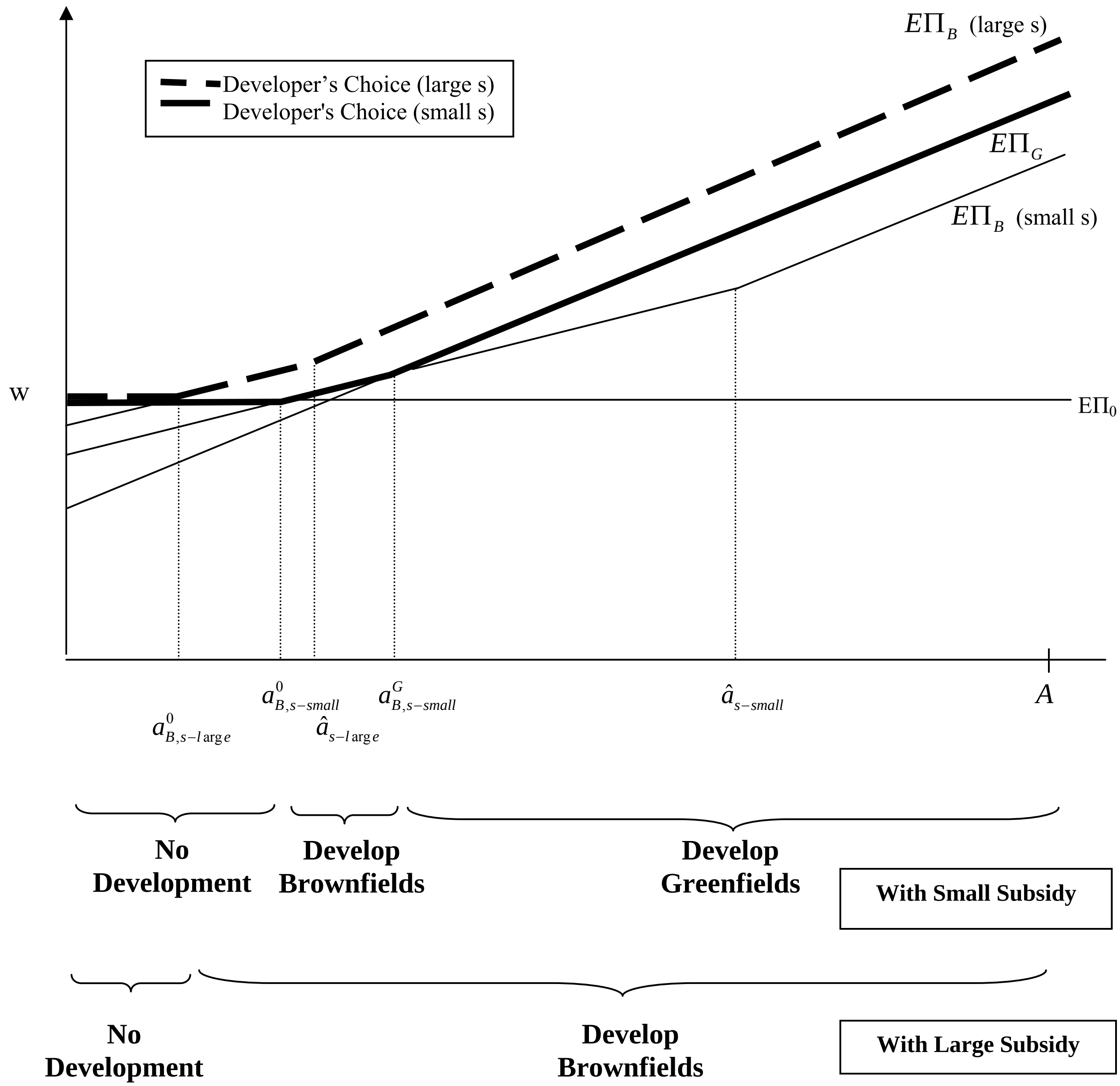
$$\frac{\partial a_B^o}{\partial s} = \frac{\partial \hat{a}}{\partial s} = -1 < 0 .$$

Thus, the subsidy increases the range of developers choosing brownfield development at both ends.[9]

The graph in Figure 2 assumes that $a_B^G < \hat{a}$. This implies that the developer who is indifferent between brownfield and greenfield development faces potential bankruptcy. However, since an increase in the subsidy both increases decreases $\hat{a}$ and increases a_B^G, eventually $a_B^G \geq \hat{a}$. This occurs when $s = r_B - r_G$ (see (9') above). At this point, the incentive for high-*a* developers to develop greenfields rather than brownfields because of adverse selection disappears. Thus, for any $s \geq r_B - r_G$, greenfield development is eliminated, and all developers, whether facing potential bankruptcy or not, decide either to develop a brownfield or not to develop at all. Figure 3 illustrates the cases for both small and large subsidies.

[9] Note that in contrast, an increase in the initial wealth *w* reduces the range of developers choosing brownfield development at both ends. This difference occurs because an increase in wealth positively affects developer returns under all development choices, while the subsidy only increases expected returns for the brownfield choice. The brownfield range shrinks because an increase in initial wealth increases the collateral exposed to bankruptcy risk.

Figure 3- Effect of an Increase in the Subsidy (s)



We now turn to the question of the efficiency implications of a subsidy. Consider first the effect of a marginal increase in the subsidy (from $s=0$). Ignoring the social cost of raising the funds to finance the subsidy, the first effect of the subsidy, namely, the windfall to developers who would have chosen brownfield development without it, has no efficiency effects. The second effect, namely, the increase in a_B^G, generates an increase in welfare by inducing some greenfield developers to switch to brownfield development. The efficiency impacts of the third effect through the decrease in a_B^o, depends on whether in the absence of the subsidy there was too much or too little development. The subsidy encourages more development at the low end. This is welfare-improving if there would otherwise be too little development, as occurs when the external benefits of brownfield redevelopment are large relative to the market impacts of potential bankruptcy. However, if the external benefits of redevelopment are small relative to the bankruptcy effects, then the market level of brownfield development is too high at the low end, implying that further encouragement to develop brownfields through a subsidy would reduce welfare in this range.

Proposition 4: *The welfare effects of providing a (small) subsidy to brownfield developers depends on the parameters, which determine the magnitudes of the external benefit from brownfield development and the market impacts of potential liability-induced bankruptcy. If* $S_B > \alpha z(L_H - L_L) - \frac{z}{1-z}w - (r_B - r)$*, then a (small) subsidy will be welfare-increasing. However, if* $S_B < \alpha z(L_H - L_L) - \frac{z}{1-z}w - (r_B - r)$ *, then the welfare effect of a (small) subsidy are ambiguous.*

Next, we ask whether it is possible to get a first-best outcome using a subsidy alone. From the above discussion, it is clear that, in order to get efficient development decisions at both ends of the distribution of *a*, the following must be true:

(13) $$s = S_B - [\alpha z(L_H - L_L) - \frac{z}{1-z}w - (r_B - r)],$$

and

(14) $$s \geq r_B - r_G \ .$$

This requires that

(15) $$S_B \geq \alpha z(L_H - L_L) - \frac{z}{1-z} w > 0 \ .$$

Proposition 5: *If the external benefit of brownfield redevelopment S_B is sufficiently high, i.e., if $S_B \geq \alpha z(L_H - L_L) - \frac{z}{1-z} w > 0$, then a subsidy given by (13) will induce a first-best outcome.*

This implies that a single instrument alone can correct the distortions at both ends of *a*. Note that the required subsidy in Proposition 5 is less than S_B since the bracketed term in (13) is negative by (11). Because, *ceteris paribus*, the bankruptcy potential encourages brownfield development, the subsidy needed to induce efficient development at the low end is less than the full external benefit of this development. However, this subsidy is more than is needed to discourage all greenfield development by high-*a* developers. To the extent that subsidies are costly to finance, this suggests that trying to correct the distortions at both ends of *a* through a single policy instrument could be unnecessarily costly. It raises the possibility that a first-best could be achieved at lower social cost through the use of two instruments instead, one to correct the distortion at the low end of *a* and another to correct the distortion at the high end of *a*. For this reason we examine the possibility of combining a subsidy for brownfield development with a tax on greenfield development.

Consider the combination of a subsidy *s* on brownfield development and a tax *t* on greenfield development. A comparison of (7) and (8') indicates that, in order for a developer with $a \geq \hat{a}$ to develop a brownfield rather than a greenfield, the following condition must hold:

$$(16) \qquad t + s \geq r_B - r_G .$$

This ensures that the cost of developing a greenfield rather than a brownfield, which consists of both the tax on greenfield development and the foregone subsidy on brownfield development, must exceed the gain from a reduced interest rate. If we set the subsidy to (13) in order to achieve efficient development at the low end of the distribution, condition (16) becomes

$$(17) \qquad t + [S_B - \alpha z(L_H - L_L) - \frac{z}{1-z} w] \geq 0 ,$$

which holds for all *t* when S_B is sufficiently high, as described in Proposition 5. When the external benefit of brownfield redevelopment S_B is sufficiently high, the subsidy can induce the efficient development outcome on both ends of the distribution by itself, and there are no social cost savings by adding a tax on greenfields. However, if $\alpha z(L_H - L_L) - \frac{z}{1-z} w > S_B > 0$, then the tax must be set at or above

$$(18) \qquad t = \alpha z(L_H - L_L) - \frac{z}{1-z} w - S_B > 0$$

in order to achieve the efficient development outcome.

Proposition 6: *If the external benefit of brownfield redevelopment S_B is not sufficiently high, i.e., if $\alpha z(L_H - L_L) - \frac{z}{1-z} w > S_B > 0$, then a subsidy given by (13) and a greenfield tax set at (18) are both necessary for a first-best outcome.*

While the subsidy can correct the inefficiency at the low end of *a*, it cannot simultaneously address the distortion at the high end. Adding a greenfield tax as described in (18) will discourage high end firms from greenfield investment, and the tax and subsidy together achieve the efficient level of brownfield investment. Note that for $S_B = 0$, this second case holds by (11), and a subsidy alone will never achieve the first-best outcome. When the difference between high and low contamination costs is large, and the brownfield externality $S_B = 0$, the market outcome yields too much brownfield investment at the low end, and even a small subsidy to encourage brownfield investment will compound the inefficiency. Therefore even if the conventional wisdom was incorrect and brownfield redevelopment did not carry with it external benefits, a simple policy instrument such as a subsidy would not be able to adequately address the market distortions created by liability-driven bankruptcy.

V. Conclusion

We develop a land choice model regarding brownfields and greenfields in a context where liability for environmental contamination on certain brownfields could potentially bankrupt some developers. We allow project returns to vary by characteristics related to the project and/or developer, and let developers sort by land preference. This allows us to observe firm development behavior when threatened by bankruptcy, both in the absence of government participation, and when influenced by government policies.

We find that brownfield investment occurs despite liability transfer and capitalization in the land price. The possibility of bankruptcy introduces a kink in expected brownfield profits, as some firms receive the discounted brownfield price, but escape bearing the full liability costs because of bankruptcy. This kink can lead to a scenario where some developers prefer greenfields, while some prefer brownfields. Firms sort over land choice based on project profitability. Furthermore, sorting will always occur in a particular order, namely; firms facing low-return (profitability) projects will not develop, firms facing medium-return projects will develop brownfields, and firms facing high-return projects will develop greenfields. This sorting can be interpreted as an adverse selection problem between firms facing medium and high-return projects, as firms facing medium-profitability projects are relieved from bearing full liability costs and so choose brownfield development, while those facing high profitability choose greenfield development.

The efficiency of the market outcome depends on the extent of the external benefit from brownfield development, and how this relates to the developer's potential liability and wealth. If the externality is relatively large enough, the market could overinvest in brownfield development. If the externality is sufficiently small relative to liability factors and wealth, the market could underinvest in brownfield development. In addition, if bankruptcy concerns lead banks to charge higher rates on brownfield development projects than on greenfield projects, then the market may overinvest in greenfield development. The market distortions caused by the brownfield development externality and the liability driven bankruptcy suggest there is a role for government intervention.

We examine the impact of a generic subsidy targeted at promoting brownfield investment. Many subsidy programs exist at the state and federal levels as either grant or loan assistance programs. We focus on subsidy programs because of their simplicity, as well as their popularity with developers. Environmental insurance programs are another potential method to address liability and bankruptcy concerns. However, as environmental insurance programs grow scarcer, enact tighter restrictions, higher premiums, lower maximum policy periods, and in general move towards more conservative underwriting, policy-makers appear more likely to use subsidy programs to support brownfield redevelopment.[10]

We find that the subsidy increases the range of brownfield developers on both the high and low end of the distribution. We show that a subsidy can correct the market and induce efficient investment on both the high and low ends if the brownfield development externality is sufficiently large. Under this scenario, there is sufficient underinvestment on the low end that a subsidy can be set to efficiently address this underinvestment, as well as prevent greenfield overinvestment due to lower greenfield loan rates. However, if the brownfield development externality is not sufficiently large, a subsidy cannot successfully address efficiency concerns, and a second instrument such as a greenfield tax is necessary to achieve the first-best outcome.

As brownfield investment can occur without government intervention, state and federal programs designed to stimulate brownfield redevelopment may encourage too much or too little brownfield redevelopment. We see that under certain circumstances, a single policy instrument is insufficient to achieve efficient investment decisions. There is a further implication of the above results. The existence of liability-driven bankruptcy means that even in a scenario where there is no brownfield development externality, a single policy instrument such as a subsidy will be insufficient to correct for the existing market investment distortions.

[10] See NKU (2006) on the trends in environmental insurance. While not the focus of this paper, large firms could alternatively create their own development portfolios with many projects, "self-insure" and thus avoid bankruptcy.

References

Alberini, Anna, A. Longo, S. Tonin, F. Trombetta and M. Turvani [2005]. "The Role of Liability, Regulation and Economic Incentives in Brownfield Remediation and Redevelopment: Evidence from Surveys of Developers," *Regional Science and Urban Economics* 35(4), 327-51.

Balkenborg, Dieter [2001]. "How Liable Should a Lender Be? The Case of Judgment-Proof Firms and Environmental Risk: Comment," *American Economic Review* 91(3): 731-8.

Bartsch, Charles and B. Dorfman [2000]. *Guide to Federal Brownfield Programs.* Northeast-Midwest Institute.

Bartsch, Charles and B. Wells [2005]. *State Brownfield Financing Tools and Strategies.* Northeast-Midwest Institute.

Boyd, James, W. Harrington, and M. Macauley [1996]. "The Effects of Environmental Liability on Industrial Real Estate Development," *Journal of Real Estate Finance and Economics* 12: 37-58.

Boyer, Marcel and J. Laffont [1997]. "Environmental Risks and Bank Liability," *European Economic Review* 41: 1427-59.

Chang, Howard F. and H. Sigman [2005]. "The Effect of Joint and Several Liability Under Superfund on Brownfields," *NBER Working Paper Series* No. 11667, September 2005.

Corona, Joel P. [2004]. *Essays in Brownfield Redevelopment*. Doctoral dissertation, University of Connecticut, 2004.

Feess, Eberhard and U. Hege [2000]. "Environmental Harm and Financial Responsibility," *The Geneva Papers on Risk and Insurance* 25(2): 220-34.

Haughney, Christine [2006]."Brownfield Lending Perks Up as Developers Eye Sites," *The Wall Street Journal Online* April 18, 2006 http://www.realestatejournal.com/propertyreport/newsandtrends/20060418-haughney.html , accessed 10/02/2006.

Heyes, Anthony G. [1996]. "Lender Penalty for Environmental Damage and the Equilibrium Cost of Capital," *Economica* 63: 311-23.

Federal Register [1995]. "CERCLA Enforcement Against Lenders and Government Entities That Acquire Property Involuntarily," *Federal Register* December 11, 1995. 60(237): 63517-9.

George Washington University (GWU) [2000]. *Public Policies and Private Decisions Affecting the Redevelopment of Brownfields: An Analysis of Critical Factors, Relative Weights and Areal Differentials.* Washington, DC. U.S. Environmental Protection Agency.

Lewis, Tracy R. and D Sappington [2001]. "How Liable Should a Lender Be? The Case of Judgment-Proof Firms and Environmental Risk: Comment," *American Economic Review* 91(3): 724-30.

Library of Congress [2002]. "Bill Number H.R.2869 of the 107th Congress," http://thomas.loc.gov/cgi-bin/query/z?c107:h.r.2869:.

National Governors Association (NGA) [2000]. *New Mission for Brownfields: Attacking Sprawl by Revitalizing Older Communities.* Washington, DC, National Governors Association.

Northern Kentucky University (NKU) [2006]. *Environmental Insurance Products Available for Brownfields Redevelopment, 2005*. Washington, DC. U.S. Environmental Protection Agency.

Pitchford, Rowan [1995]. "How Liable Should a Lender Be? The Case of Judgment-Proof Firms and Environmental Risk," *American Economic Review* 85(5): 1171-86.

Pitchford, Rowan [2001]. "How Liable Should a Lender Be? The Case of Judgment-Proof Firms and Environmental Risk: Reply," *American Economic Review* 91(3): 739-45.

Schwarz, Peter and Alex Hanning [2005]. "Reducing Liability to Stimulate Brownfield Redevelopment: Model and Simulation," paper presented at the Southern Economic Association annual meeting, Washington, DC November 2005.

Segerson, Kathleen [1993]. "Liability Transfers: An Economic Assessment of Buyer and Lender Liability," *Journal of Environmental Economics and Management* 25: S-46-S-63.

Segerson, Kathleen [1994]. "Property Transfers and Environmental Pollution: Incentive Effects of Alternative Policies," *Land Economics* 70(3): 261-72.

Segerson, Kathleen [1997]. "Legal Liability as an Environmental Policy Tool: Some Implications for Land Markets," *Journal of Real Estate Finance and Economics* 15(2): 143-59.

Stiglitz, Joseph E. and A. Weiss [1981]. "Credit Rationing in Markets with Imperfect Information," *American Economic Review* 71(3): 393-410.

U.S. Conference of Mayors (USCM) [2000]. *Recycling America's Land: A National Report on Brownfield Redevelopment.* Washington DC, US Conference of Mayors Brownfield Taskforce.

U.S. Environmental Protection Agency (EPA) [2006]. http://www.epa.gov/brownfields , accessed 10/02/2006.

U.S. Environmental Protection Agency (EPA) [2006]. “Summary of the Small Business Liability Relief and Brownfields Revitalization Act,” http://www.epa.gov/swerosps/bf/html-doc/2869sum.htm , accessed 10/02/2006.

Wernstedt, Kris, P.B. Meyer, A. Alberini, and L. Heberle [2006]. “Incentives for Private Residential Brownfields Development in US Urban Areas,” *Journal of Environmental Planning and Management* 49(1): 101-19.

White House [2002]. www.whitehouse.gov/news/releases/2002/01/print/20020111-3.html , accessed 10/02/2006.

Comments on the papers by Sigman, Corona and Segersen, and Schwarz and Hanning

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While reviewing these three papers I was reminded of the old joke about the three statisticians on a hunting trip. I'm sure you've heard it. They sight a magnificent deer; the first one raises his gun, fires, and misses a meter to the left. The second raises her gun, fires, and misses a meter to the right. The third puts his gun down, pumps his fist in the air, and shouts "We got it!"

I am, of course, trying to be funny. These papers are all serious attempts to better understand important phenomena and, I think, offer some important insights. I wouldn't call any a "miss". The grain of truth that brought the old joke to mind, though, is that if I were to say what might be done on the margin to improve their already useful analyses, it would be to nudge them a little closer to one another. Hilary Sigman's excellent paper on the consequences of joint and several liability might be a little more complete if Hilary offered the reader a little more help with theory to interpret her compelling empirical results. Joel Corona and Kathleen Segerson's work on liability and the choice of development location might benefit from a few more details relating their results to real-world policy environments and, I think, sets the stage for some interesting empirical questions. Finally, I believe that Peter Schwarz and Alex Hanning's paper, while it raises some intriguing possibilities, might be improved both by providing more empirical foundation and considering a broader range of potential explanations for the phenomenon they investigate.

The most striking features of Hilary Sigman's paper are that legal standards imposing joint and several liability on potentially responsible parties depress property values and inhibit transactions in potentially contaminated lands. Perhaps this is the sort of thing that strikes normal people and economists differently (yes, I do intend that to sound ironic, and was amused at Hilary's own description of the different way she thinks of things "as a person" as opposed to "as an econometrician"). If you told the proverbial person on the street that imposing joint and several liability reduces the demand for property, the reaction might be (at least among those who recognized the terms) "D'oh!" On first inspection, it's the sort of thing that ought to be true.

On closer inspection, though, the issue is not "should contaminated land sell for less?" Of course it should. The point, rather, is "given that the market knows – or at least suspects – that land is contaminated, why and how would the nature of liability imposed for land contamination affect transactions in land?"

I'd like to see Hilary's paper take a little more time addressing these matters in its current sections 1.1 and 1.2, as well as in the conclusions. She provides a synopsis of the issues, summarizing informational asymmetries, judgment proof parties, etc., but I was wondering as I read if there might not be another layer of complexity to this. To me, it seems that liability, in whatever form, ought to amenable to analysis by the Coase Theorem: there should be efficient ways to allocate the burden of liability, and the efficiency of the outcome ought not to depend on the identity of the payer. To the extent that joint and several liability really reduces property values, then, I was wondering if it's not because joint and several liability results in escalation of transactions costs in such a way as to diminish the overall surplus afforded by property ownership. It might be helpful to walk the reader through these issues.

In this sense, perhaps Hilary's paper could borrow an expositional feature or two from Peter Schwarz and Alex Hanning's. Perhaps I'm being a little too simplistic here, but we might separate a prospective buyer's willingness to pay among value-in-use, potential liability, and potential transactions costs arising in the assignment of liability. The last of these is a pure social loss, and it might be useful to trace it through to who in the transaction pays the transaction cost (although it would seem to me to be another corollary of the first theorem of public finance: incidence is unrelated to who signs the check).

I had a sort of parallel reaction to Hilary's discussion of negligence vs. strict liability. It surprised me a little that Hilary finds that strict liability doesn't affect transactions price relative to negligence. Her explanation is reasonable: to paraphrase Camus, "we are all negligent". Meeting the standard may not absolve an owner of liability. I wondered about this, though. *Ideally*, owners would incur equal expenses under both strict liability and negligence standards, and the price of a property would differ under the two regimes by the expected present value of the liability incurred under the strict regime. But what if the negligence standard is not effective, and an owner might escape some liability by limited capitalization, or, as in some recent work by Andrei Shleifer and others, subverting the legal process? I'm not sure how plausible this may be, but rather, am only suggesting that there might be some alternative explanations of the phenomenon Hilary identifies.

I'll note in concluding my remarks on Hilary's paper first, that the most interesting finding in Hilary's paper may not involve prices, but rather, vacancies. As asymmetric information models might predict, the real problem may be that transactions don't occur when they should. Finally, while I suppose any test of endogeneity can only be as good as the instruments proposed, Hilary seems to have followed best practice, and, since she cannot reject exogeneity, creates a reasonable presumption that her results and their interpretation are valid: joint and several liability induces less efficient transations markets, rather than intrinsic features of local markets inducing adoption of joint and several liability.

Joel Corona and Kathleen Segerson's paper is another in an interesting series of papers on the distribution of the burden of liability and the efficiency of property markets. I'm not entirely confident that I'm going to get this right, but I *think* the basic idea is something like the following. Property developers differ in their abilities. Some guys know their prospects aren't great. The outcome of their development activities depends on two things: their own (unobservable save to themselves) intrinsic ability, and the likelihood that contamination is identified in the property they are developing. Guys who know they're intrinsically less able like the riskier option of developing brownfields, since there's a greater likelihood that they can wash their hands of the project if it doesn't go well. Conversely, guys who know themselves to be possessed of high ability will prefer to develop in greenfields, since they know they won't be dispossessed of the fruits of their labors by a bad realization of contamination conditions.

Two aspects Joel and Kathleen's exposition troubled me. The first is simply complexity. By my count, 23 different variables were introduced in the analysis (you can count if you like: a, $\hat{a}$, a^*, a_B^o, a_B^G, α, L_L, L_H, P_B, P_G, π_B, π_G, r, r_B, s, S_B, S_G, t, V_o, $V_D^B(a)$, $V_D^G(a)$, w, z). Entire languages get by with smaller alphabets! I suppose that, as Einstein said, everything should be made as simple as possible, but no more so. Still, I did find the expressions tough sledding, and would have appreciated a little more guidance and intuition along the way.

The second thing that troubled me was that I wasn't quite sure what conditions would need to obtain in the credit supply side of the market. Joel and Kathleen proceed from the reasonable assumption that borrowers whose prospects are riskier will need to pay higher interest rates (in states of the world in which they don't default, that is), but I don't believe that they explicitly impose conditions such as zero-expected-profits on lenders to motivate their findings. I'm certainly not asking for more notation (!), but again, would appreciate some greater guidance on how the model is being developed, and justification of conditions that didn't seem to me to be entirely firmly established.

[In a later conversation with Joel he told me that an earlier version of the paper included more details on the credit supply side of the market. This comes as no surprise, as the modeling work in generally seems to be well and thoroughly done. It might be wise to consign such details to technical appendices and present mostly intuitions and diagrams in the main text.]

I'd also be interested in knowing what the stylized facts of land development and redevelopment markets are. It seems that Joel and Kathleen are deriving testable implications – developers operating in brownfields ought to default more often – and I'd be interested in seeing the evidence of this. The paper is motivated by an important policy concern: is there too much or too little brownfield development? It would be useful to know, though, if the model proposed to answer that question is also consistent with the stylized facts of developer performance.

These stylized facts would also be useful as they'd establish whether the phenomenon Joel and Kathleen model is important in practice or if it might be obviated (as I suspect, but cannot prove, it is) by other practices, such as requiring brownfield developers to demonstrate adequate capitalization to avoid bankruptcy, etc. I was intrigued by Joel's remark during his (generally quite clear and nicely motivated, by the way) presentation that some firms appear to specialize in brownfield development. There could well be technical reasons for this (specialized knowledge would help, I'm sure), but Joel and Kathleen's analysis also begs the question of whether firms might specialize for reputational or capitalization reasons. I should add that while such other practices might "obviate" the specific form of differential credit-market pricing Joel and Kathleen describe, it would by no means obviate their larger point that asymmetries of information could motivate inefficiencies.

The paper by Peter Schwarz and Alex Hanning is, I think, the most schematic of the three, and seems to me to leave the most loose ends. The basic idea seems to be pretty straightforward: if the regulatory regime affords one party greater immunity from liability than another, ownership will come to be assigned to the party for whom the net of value-in-use less liability is greatest. It is, then, entirely possible that someone who cannot use a property as efficiently as can another will end up controlling it simply because he cannot be held to account for damage as fully as can another.

At one level, this is an interesting observation. The whole market-for-lemons problem is an *efficiency* problem only to the extent that would-be sellers with private information are unable to use land as profitably as could would-be buyers.

Peter and Alex make a good point on equity, though: there is surely some social opposition to "giving polluters a break".

I think, though, that Peter and Alex's paper is incomplete, as its analysis begs several questions. First, one wonders about the possibility of sham "ownership" to avoid liability. If, simply by changing the name on the deed, one could escape a substantial share of liability, wouldn't there be very strong incentives to do so, while maintaining the same management and effective control of the property? [I'm reminded of what I think is a not entirely inappropriate analogy: when I was in high school in Washington State the Boldt native-rights ruling determined that Native Americans were entitled to a large share of the northwest salmon run. Very shortly thereafter, at least according to my high-school friends and their families engaged in the fishing business, several local boats were "acquired" by Native skippers.]

Second, I wondered as I read if there might not be more to the story. Why should a change in ownership occasion a change in the assignment of liability? Might there be some aspect of the fact of a transaction occurring at all that would motivate a change in liability? In short, there's been a lot of discussion in the literature of the role of asymmetric information, and it seems that one would want to weave those issues into the story Peter and Alex tell.

Finally, one would like to have some better sense of the empirical importance of the phenomena being discussed. Granted, it's next-to-impossible to know how different parties differ in their ability to exploit the potential of a property, but the paper does make the reader wonder how important these differential-assignation-of-liability concerns are likely to be in practice.

Comments on the Sigman, Corona-Segerson and Schwarz-Henning papers

By

Anna Alberini
17 October 2006

What factors influence brownfield transactions and reuse? These three papers study the effects of liability (Sigman), subsidies to brownfield redevelopment and taxes on greenfield development (Corona-Segerson), and risk-based cleanup standards (Schwarz) on the price, sale and reuse of potentially contaminated sites. They thus collectively span a wide range of policies that are currently used or under consideration to encourage cleanup and redevelopment of BF properties, and they are very well suited for being presented as part of a workshop session. Another appeal of this session is the fact that there is one empirical paper (Sigman) and that the other two papers are theoretical but generate a number of hypotheses that might eventually be empirically tested.

Liability is the focus of Sigman's paper, which could be used to predict the effects of relaxing the liability imposed on owners of contaminated property, and figures prominently in the other two papers, which assume liability to exist, but do not get into the specifics of the liability regime. The three papers rely on earlier theoretical development by Segerson (1993, 1994), who shows that as long as all parties are solvent, contaminated properties will be sold and bought, and that is does not matter whether liability is imposed on the buyer or the seller. When one of the parties is insolvent, however, liability may, under certain conditions, discourage transactions. Boyd et al. (1996) focus on informational asymmetries between sellers and buyers, and between property owners and the government. Such informational asymmetries can, under certain conditions, deter sales of potentially contaminated properties.

In what follows I discuss the papers individually.

Sigman paper.

General comments. The reasoning behind this paper is more stringent liability regimes raise the cost of redeveloping potentially contaminated property, encouraging developers to seek pristine properties ("greenfields") where liability issues would not arise. The end result is that the price of greenfields rises relative to that of brownfields, and the latter remain unsold and/or underutilized.

Using data from the Society of Industrial and Office Realtors (SIOR), which document industrial real estate, Sigman focuses on a system of two (unrelated) equations, where the dependent variables are (i) property prices, and (ii) vacancy rates.[1] For most of the cities covered by the data, (i) and (ii) are available for both city center and for suburban areas,

[1] Sigman also uses data reported from the US Conference of Mayors. However, this appears to be a less reliable dataset that also spans for less years than the SIOR data, so my comments focus on the latter.

making it possible to do matched-data analyses. The regressions control for employment, former and present industrial activity in the area, etc. they also include—and these regressors are at the heart of the paper—dummies for whether the state mini-Superfund program imposes for strict and/or joint-and-several liability.

I concur with Sigman that these are the appropriate liability incentives for properties with low-to-medium contaminations, which would not be falling on the shoulders of the federal Superfund program. The finding that these features of the state programs are not simultaneously determined with the outcomes being studied are also consistent with my expectations—the state mini-Superfund statutes were presumably crafted to address other problems, rather than the lack of transactions at BFs.

I was, however, surprised that strict liability is not a significant determinant of (relative) prices and vacancy rates, while joint-and-several liability is. I was especially surprised by the magnitude of the latter effect, even when (or especially because) due consideration is given to the fact that vacant properties account for a very small fraction of the stock of properties slated for industrial use.

I was expecting strict v. negligent-based liability to matter because, together by enforcement effort, this feature determines the extent of expected cleanup costs. The best conjecture I can offer for this non-result is that perhaps by the beginning of the period covered by the data, anyone who wanted to buy industrial property was having their due diligence assessments and inspections done before closing (or turning down) deals.

I did read the Chang and Sigman (2005) paper, which proposes theoretical models that explain why joint-and-several liability deter transactions at potentially contaminated properties. However, I would expect joint-and-several liability to matter when there are multiple parties involved at a site, and they are uneven in terms of solvency and net assets. Is this always the case with industrial properties? What happens, for example, when contaminated properties changes hands because of corporate mergers or takeovers? Does something like this account for a large share of transactions? Are most transactions really as assumed by the theoretical literature, i.e., the buyer has deeper pockets than the seller?

Another possible explanation for the large effect of joint-and-several liability is that this is an econometric artifact—in other words, that the joint-and-several liability captures something else that is correlated with it (and that was not entirely captured into the city-specific fixed effects).

A suggestion. I am not convinced that the paper was able to capture (or refute) substitution of greenfields for brownfields. I would recommend that Sigman tries an explicit test of this hypothesis as outlined below. This empirical test follows intuitively from the notion of substitution, which is inherently a dynamic concept. Specifically, estimate the regression equation:

(1) $$\Delta GF_{it} = \gamma_0 + \gamma_1 \cdot \Delta BF_{it} + (\text{other variables}) + \eta_{it}$$

where ΔGF_{it} and ΔBF_{it} are the *changes* of GF and BF vacancy rates (or better yet, sales or [re]developed land) from one period to the next. The sign of coefficient γ_1 will tell us if there is substitution between GFs and BFs (if γ_1 is negative) or if GFs and BFs are complementary (if γ_1 is positive).

Policy implications. Because two separate strict- and joint-and-several liability dummies are entered in the RHS of the regression equations, the regressions of this paper assume symmetric effects of imposing/removing liability. Is it really so? Did you try a model that includes an interaction between these two dummies? Is it possible that having had liability in the past changes permanently behaviors and expectations, almost as a "duration dependence" type of effect?

Other econometric issues.

- I believe that much of the variation in the liability regimes is across states, rather than within states over time. But the 'within' estimator (the estimator appropriate for models with fixed effects) ignores the difference between one state and the sample averages, focusing solely on variation over time within states.
- I believe cities with strong economies and hot real estate markets are less sensitive to the incentives of liability. Did you try running your regressions without these cities? Also, include controls for the strength of the economy interacted with the liability dummies (not just entered additively)?
- Did you control for other policies that might have affected the outcomes (e.g., EZ designations and funding, BF designations and funding, etc.)?
- The paper is silent on the fact that industrial properties are a rather "difficult" market to work. Jackson (2002) points out that only recently have these property begun selling—how does that affect the quality of the your data and any trends that they might exhibit over time?
- Is it possible to look at other outcomes instead of vacancy rates (e.g., turnover rates, number of sales, number of non-arms-length sales)?

Corona-Segerson paper

This paper presents a simple and elegant framework for analyzing sales of brownfields in the presence of liability, and for studying the effects of BF subsidies (and of GF taxes).

One problem with this paper is that it is really heavy on the notation, and that I was under the impression that "stock" type of quantities (e.g., assets) got mixed up with "flow" type of quantities (e.g., profits from the development project). For example, if *w* represents assets, why would the expected profits from undertaking no development project at all be written as $E\pi_0 = w$? Shouldn't it be $E\pi_0 = wr$, where *r* is the interest rate associated with any other use of capital? And, what is the relationship between the asssets of the firm and the collateral used for securing loans? Perhaps this does not make a difference in terms of the main gist of the paper, but I definitely got confused.

I was also having a hard time separating project types from developers type, and any clarification that might help the faint of heart would be appreciated.

The paper would be even more compelling if it was possible to incorporate other factors that observers typically linked with abandoned contaminated properties:

- Transaction costs
- The cost of acquiring information about the true contamination at a property (e.g., environmental assessment) and the effects of a subsidy specific for this type of activity. This would have great policy relevance, because the EPA Brownfield Programs does offer environmental assessment grants.
- Uncertainty about the real estate market, the prices for which properties can be sold, and ultimately about profits. In weak markets, BF development may do worse than in a strong market, and the policy mix necessary to stimulate reuse might change. (By contrast, uncertainty about the extent of contamination and cleanup costs is well addressed by this paper.)
- Does the model accommodate situations where the subsidies are proportional to the cleanup costs, as is the case with the Brownfield Tax Credit?

Schwarz-Henning paper

Does economic theory confirm expectations that relaxing cleanup standards and linking them to land use and exposure should encourage sales of contaminated sites? This paper shows how the extent to which this is so depends on whether risk-based cleanup standards are allowed only for the buyer, or to both seller and buyer.

One thing that is missing from this paper is a clear discussion of current policy in terms of risk-based cleanup standards. For example, I believe that during the 1990s most UST programs in the various states adopted risk-based corrective action. Can we observe some of the predictions of the Schwarz-Henning paper within these programs?

Market Mechanisms and Incentives: Applications to Environmental Policy

A Workshop Sponsored by the U.S. Environmental Protection Agency's National Center for Environmental Economics (NCEE) and National Center for Environmental Research (NCER)

Resources for the Future
1616 P Street, NW, Washington, DC 20036
October 17-18, 2006

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Market Mechanisms and Incentives: Applications to Environmental Policy

Resources for the Future
1616 P Street, NW, Washington, DC 20036
(202) 328-5000

October 17th – 18th, 2006

October 17, 2006: Market Mechanisms in Environmental Policy

8:00 a.m. – 8:45 a.m. **Registration**

8:45 a.m. – 11:45 a.m. **Session I: Brownfields and Land Issues**
Session Moderator: **Robin Jenkins**, EPA, National Center for Environmental Economics

8:45 a.m. – 9:00 a.m. Introductory Remarks: **Sven-Erik Kaiser**, EPA, Office of Brownfields Cleanup and Redevelopment

9:00 a.m. – 9:30 a.m. Environmental Liability and Redevelopment of Old Industrial Land
Hilary Sigman, Rutgers University

9:30 a.m. – 10:00 a.m. Incentives for Brownfield Redevelopment: Model and Simulation
Peter Schwarz and **Alex Hanning**, University of North Carolina at Charlotte

10:00 a.m. – 10:15 a.m. **Break**

10:15 a.m. – 10:45 a.m. Brownfield Redevelopment Under the Threat of Bankruptcy
Joel Corona, EPA, Office of Water, and **Kathleen Segerson**, University of Connecticut

10:45 a.m. – 11:00 a.m. Discussant: **David Simpson**, EPA, National Center for Environmental Economics

11:00 a.m. – 11:15 a.m. Discussant: **Anna Alberini**, University of Maryland

11:15 a.m. – 11:45 a.m. Questions and Discussion

11:45 a.m. – 12:45 p.m. **Lunch**

12:45 p.m. – 2:45 p.m. **Session II: New Designs for Incentive-Based Mechanisms for Controlling Air Pollution**
Session Moderator: **Will Wheeler**, EPA, National Center for Economic Research

12:45 p.m. – 1:15 p.m. Dynamic Adjustment to Incentive-Based Environmental Policy To Improve Efficiency and Performance
Dallas Burtraw, Danny Kahn, and Karen Palmer, Resources for the Future

	1:15 p.m. – 1:45 p.m.	Output-Based Allocation of Emissions Permits for Mitigating Tax and Trade Interactions **Carolyn Fischer**, Resources for the Future
	1:45 p.m. – 2:00 p.m.	Discussant: **Ann Wolverton**, EPA, National Center for Environmental Economics
	2:00 p.m. – 2:15 p.m.	Discussant: **Arik Levinson**, Georgetown University
	2:15 p.m. – 2:45 p.m.	Questions and Discussion
2:45 p.m. – 3:00 p.m.	**Break**	
3:00 p.m. – 5:30 p.m.	**Session III: Mobile Sources** Session Moderator: **Elizabeth Kopits**, EPA, National Center for Environmental Economics	
	3:00 p.m. – 3:30 p.m.	Tradable Fuel Economy Credits: Competition and Oligopoly **Jonathan Rubin**, University of Maine; **Paul Leiby**, Environmental Sciences Division, Oak Ridge National Laboratory; and **David Greene**, Oak Ridge National Laboratory
	3:30 p.m. – 4:00 p.m.	Do Eco-Communication Strategies Reduce Energy Use and Emissions from Light Duty Vehicles? **Mario Teisl**, **Jonathan Rubin**, and **Caroline L. Noblet**, University of Maine
	4:00 p.m. – 4:30 p.m.	Vehicle Choices, Miles Driven, and Pollution Policies **Don Fullerton**, **Ye Feng**, and **Li Gan**, University of Texas at Austin
	4:30 p.m. – 4:45 p.m.	Discussant: **Ed Coe**, EPA, Office of Transportation and Air Quality
	4:45 p.m. – 5:00 p.m.	Discussant: **Winston Harrington**, Resources for the Future
	5:00 p.m. – 5:30 p.m.	Questions and Discussion
5:30 p.m.	**Adjournment**	

October 18, 2006:

8:45 a.m. – 9:15 a.m.	**Registration**	
9:15 a.m. – 12:20 p.m.	**Session IV: Air Issues** Session Moderator: **Elaine Frey**, EPA, National Center for Environmental Economics	
	9:15 a.m. – 9:45 a.m.	Testing for Dynamic Efficiency of the Sulfur Dioxide Allowance Market **Gloria Helfand**, **Michael Moore**, and **Yimin Liu**, University of Michigan
	9:45 a.m. – 10:05 a.m.	When To Pollute, When To Abate: Evidence on Intertemporal Use of Pollution Permits in the Los Angeles NOx Market **Michael Moore** and **Stephen P. Holland**, University of Michigan

10:05 a.m. – 10:20 a.m.	**Break**	
	10:20 a.m. – 10:50 a.m.	A Spatial Analysis of the Consequences of the SO_2 Trading Program **Ron Shadbegian**, University of Massachusetts at Dartmouth; Wayne Gray, Clark University; and Cynthia Morgan, EPA
	10:50 a.m. – 11:20 a.m.	Emissions Trading, Electricity Industry Restructuring, and Investment in Pollution Abatement **Meredith Fowlie**, University of Michigan
	11:20 a.m. – 11:35 a.m.	Discussant: **Sam Napolitano**, EPA, Clean Air Markets Division
	11:35 a.m. – 11:50 a.m.	Discussant: **Nat Keohane**, Yale University
	11:50 a.m. – 12:20 p.m.	Questions and Discussion
12:20 p.m. – 1:30 p.m.	**Lunch**	
1:30 p.m. – 4:35 p.m.	**Session V: Water Issues** Session Moderator: **Cynthia Morgan**, EPA, National Center for Environmental Economics	
	1:30 p.m. – 2:00 p.m.	An Experimental Exploration of Voluntary Mechanisms to Reduce Non-Point Source Water Pollution With a Background Threat of Regulation Jordan Suter, Cornell University, Kathleen Segerson, University of Connecticut, Christian Vossler, University of Tennessee, and Greg Poe, Cornell University
	2:00 p.m. – 2:30 p.m.	Choice Experiments to Assess Farmers' Willingness to Participate in a Water Quality Trading Market **Jeff Peterson**, Washington State University, and Sean Fox, John Leatherman, and Craig Smith, Kansas State University
2:30 p.m. – 2:45 p.m.	**Break**	
	2:45 p.m. – 3:15 p.m.	Incorporating Wetlands in Water Quality Trading Programs: Economic and Ecological Considerations **Hale Thurston** and Matthew Heberling, EPA, National Risk Management Research Laboratory, Cincinnati, Ohio
	3:15 p.m. – 3:35 p.m.	Designing Incentives for Private Maintenance and Restoration of Coastal Wetlands **Richard Kazmierczak** and **Walter Keithly**, Louisiana State University at Baton Rouge
	3:35 p.m. – 3:50 p.m.	Discussant: **Marc Ribaudo**, USDA, Economic Research Service
	3:50 p.m. – 4:05 p.m.	Discussant: **Jim Shortle**, Pennsylvania State University
	4:05 p.m. – 4:35 p.m.	Questions and Discussions
4:35 p.m. – 4:45 p.m.	**Final Remarks**	

4:45 p.m. **Adjournment**

Dynamic Adjustment to Incentive-Based Policy to Improve Efficiency and Performance

Dallas Burtraw, Danny Kahn and Karen Palmer
Resources for the Future

November 29, 2006

<< Work in Progress – Comments Welcome >>

Burtraw and Palmer are senior fellows and Kahn is a research assistant at Resources for the Future. This research was presented at the EPA National Center for Environmental Research (NCER) Progress Review Workshop on Market Mechanisms and Incentives, October 17-18, 2006 in Washington DC. This research was funded in part by EPA Agreement Number RD 83099001. Send comments to <author's last name>@RFF.org.

Dynamic Adjustment to Incentive-Based Policy to Improve Efficiency and Performance

Dallas Burtraw, Danny Kahn and Karen Palmer

Abstract

The issue of how to set policy in the presence of uncertainty has been central in debates over climate policy, where meaningful efforts to control emissions could prove much more costly than prior regulatory efforts to limit emissions of air pollution. Concern about costs has motivated the proposal for a cap and trade program for carbon dioxide, with a provision called a "safety valve" that would mitigate against spikes in the cost of emission reductions by introducing additional emission allowances into the market when marginal costs rise above the specified allowance price level.

We find two significant problems with this approach, both stemming from the asymmetry of an instrument that mitigates *only* against a price *increase*. One is that the most important examples of price volatility in cap and trade programs have occurred not when prices spiked, but instead when allowance prices fell below their expected values. For example, in the case of SO_2 emission trading, the inability of the trading program to adjust to the fall in allowance prices led to welfare losses of between \$1.5 and \$8 billion dollars per year, measured against congressional intent. A second problem is that a single-sided safety valve may affect the behavior of investors with unintended consequences. Using a Taylor series expansion around the mid case for uncertain future natural gas prices, we measure the distribution of key variables of interest in a detailed electricity market model, where we show that a high-side safety valve can be expected to increase emissions and decrease investment in nonemitting technologies, relative to the absence of a safety valve altogether.

A symmetric safety valve is a price stabilization policy that addresses both unanticipated spikes or drops in the allowance price. When allowance price falls below the safety valve floor, the symmetric safety valve contracts the number of allowances issued in the market. In simulation analysis with uncertain natural gas prices, the symmetric safety valve returns the expected value for these and other key parameters to near their levels in the absence of a safety valve. In addition, although a high-side safety valve improves welfare, a symmetric safety valve improves welfare even further. In summary, we find a symmetric safety valve can improve the performance of allowance trading programs, improve welfare, and may help overcome political objections from environmental advocates who have opposed the use of a safety valve.

Key Words: emission allowance trading, climate change, market-based policies

JEL Classification Numbers: Q4, L94, L5

Dynamic Adjustment to Incentive-Based Policy to Improve Efficiency and Performance

Contents

Dynamic Adjustment to Incentive-Based Policy to Improve Efficiency and Performance

Dallas Burtraw, Danny Kahn and Karen Palmer

1. Introduction

Policymakers advance economic efficiency when they set policy goals at levels that equate the marginal costs of additional pollution controls with the marginal benefits of improvements in environmental quality and when they employ incentive-based approaches, such as tradable permits or taxes, to achieve these goals in a least cost manner. When attempting to set goals, policymakers face a great deal of uncertainty about the costs and benefits to society of achieving a particular goal and, in particular, how those costs and benefits are likely to change over time. The presence of uncertainty about both the costs and benefits of regulation also effects on the choice of policy instruments (Weitzman 1974, Roberts and Spence 1978, Pizer 2002).

The issue of how to set policy in the presence of uncertainty has been particularly salient in debates over climate policy, where meaningful efforts to control emissions could prove much more costly than prior regulatory efforts to limit emissions of air pollution. Both the costs and benefits of controlling emissions of greenhouse gases are highly uncertain. One proposal that helps to neutralize this potential would be to include in a carbon cap and trade program a “safety valve” that serves as a ceiling on the price of carbon emission allowances by increasing the provision of emission allowances in the market (Pizer 2002; Kopp et al. 2002). This proposal has found favor with some federal policy makers in the United States and is incorporated in the climate policy section of the comprehensive energy policy advanced by the National Center for Energy Policy in 2004 and later incorporated into legislative language by Senator Bingaman (D-NM), although not yet formally introduced in the Senate. The safety valve cap on allowance prices is also a feature of a climate cap and trade legislative proposal introduced in the House of Representatives by Representatives Udall (D-NM) and Petri (R-WI). A safety valve provision also was incorporated in the proposed Clear Skies Act that would have imposed national caps on emissions of SO_2, NO_x and mercury from electricity generators.

To date most advocates of the safety valve approach focus exclusively on the situation where realized costs of reducing pollution turn out to be higher than expected and thus the original emissions cap is no longer efficient. However, evidence is that *ex post* actual costs of government regulation are often lower than *ex ante* expected costs (Harrington et al. 2000). In many cases, total costs turn out to be lower than expected because baseline emission levels were overestimated and the emissions reductions necessary to achieve a target level end up being less than originally anticipated. Thus since total costs and aggregate reductions tend to be overestimated, the *ex ante* estimates of unit costs (e.g. cost per ton of emissions reduced) tend to be more consistent with *ex post* experience than do estimates of total costs. However, in virtually every case when an incentive form of regulation like an emissions cap and trade approach is used, unit costs have been overestimated prior to the regulation taking effect (Harrington et al. 2000). Indeed, the economic benefits of insuring against the prospect of costs that are *lower* than expected appear at least as important as the benefits of insuring against costs that are *higher* than expected, based on experience with cap and trade programs to date. A key example is the SO_2 caps under Title IV of the 1990 Clean Air Act Amendments. We calculate that a safety valve protecting legislative intent against the prospect that costs would be substantially lower than expected would have improved economic welfare by $1.5 billion to $8.25 billion per year.

The possibility that costs may turn out to be lower than expected or, in a related vein that benefits may turn out to be higher than expected, suggests that the single-sided safety valve that serves to cap the high-side of the allowance price does not provide sufficient insurance against uncertainty. In particular, if costs turn out to be much lower than expected, or if benefits are much higher, the emission cap set forth in a regulation or a piece of legislation could prove dramatically insufficient and also, in many cases, very difficult to change. A regulatory design that could help improve the efficiency of policies in this situation would be a double-sided, or *symmetric* safety valve that sets both a floor and a ceiling on the price of emission allowances.

Using a detailed simulation model of the electricity sector, we find a more pervasive consequence of a single-sided safety valve should be expected. The safety valve affects the expectations of allowance prices, and thereby affects the expectations about the payoff from various investment strategies. Accounting for uncertainty in the future price of natural gas, we find that the single-sided safety valve is likely to reduce the investment in nonemitting technology and increase the expected emissions that obtain under the policy. However, a symmetric safety valve with a floor as well as a ceiling on the price of emission allowances recovers the expected payoff to investments in nonemitting technologies as well as the expected environmental performance of the program. In so doing, it is likely to repair the political coalitions that have been somewhat fractured by the discussion of safety valves to date. A symmetric safety valve would preserve all of the virtue while avoiding the unfortunate unintended consequences of a single-sided approach.

2. Literature Review

The literature addressing the question of instrument choice for environmental policy in the presence of uncertainty about the costs and/or benefits of regulation extends back over thirty years. Early work by Weitzman (1974) identifies conditions under which price instruments would be preferable to quantity instruments and vice versa. In his model, which is the more efficient instrument hinges on the relative slopes of the marginal benefit and marginal cost curves. He shows that quantity instruments are preferred to price instruments if the marginal benefits curve is steeper than the marginal cost curve. In a subsequent paper, Roberts and Spence (1978) analyze a combination of a quantity constraint and a price instrument, which is specified as a licensed target level of emissions, a per unit subsidy for reductions in emissions below the firm's licensed target level and a penalty for emissions above the target where the penalty is weakly greater than the subsidy. Roberts and Spence prove that this hybrid approach yields a lower value of total social costs (defined as the sum of pollution damages and clean-up costs) than would result from using either instrument in isolation. Note that if damages are linear, then the pure tax is optimal, and (1978) finds a similar result when damages are linear. Pizer (2002) uses a computable general equilibrium simulation model to analyze the welfare consequences of using different instruments to reduce CO_2 emissions that contribute to climate change. His work shows that the expected welfare gains from a price approach to climate policy are 5 times higher than expected gains with a pure quantity approach. The optimal hybrid policy of a CO_2 emissions cap coupled with a cap on the price of CO_2 allowances yields slightly higher net social benefits than a tax policy by itself, and also substantially outperforms the pure quantity approach. The hybrid approach outperforms the pure price policy because, in this case, the climate benefits function is slightly convex.

Pizer does not consider a lower bound on the price of allowances and dismisses the use of a floor on allowance prices, implemented in the form of a commitment to buy back allowances once the price falls to the level of the floor, as having adverse dynamic properties as discussed in Chapter 14 of Baumol and Oates (1988). They argue that subsidizing emissions reductions in a competitive industry will typically lead to decreased output at the firm level, but increased output at the industry level and can also lead to higher emissions. They also cite Wenders (1975) who argues that subsidizing emission reductions can reduce incentives for the adoption of a new pollution-reducing innovation if firms anticipate that adopting the new technology will reduce subsidy payments. In both these cases, the assumption is that firms have the property rights to emissions and the government is buying them back. However, the symmetric safety valve need not be implemented as a government buy-back of allowances that were previously distributed for free. For example, if some portion of allowances is being sold in an auction instead of distributed gratis, the low-side safety valve could take the form of a floor on the price of those auctioned allowances. If the willingness to pay for emission allowances were to fall below that floor, less than the allowable quantity of emission allowances should be sold. This is a standard practice in auctions when willingness to pay is uncertain (Burtraw and Palmer, 2006). Also, if compliance periods extend for multiple years and low prices prevail during the early years, allocations in the later years of the compliance period could be reduced, which would lead to higher prices in earlier years if allowance banking is allowed and in effect.

An important insight from the Weitzman (1974) paper, which has typically been ignored in subsequent work, is the role of correlation of benefits and costs in the identification of optimal instruments. Stavins (1996) shows that when benefits and costs are statistically correlated, benefit uncertainty can affect instrument choice and the extent of that effect depends on several parameters. When benefits and costs are positively correlated, a quantity approach to regulation tends to be preferred to a price approach and when the correlation is negative, the tax approach will tend to be preferred. Stavins argues that positive correlation is more likely and that in general correlation in benefits and costs tends to favor emissions caps over emissions taxes. Evans (2006) considers correlation among the cost of control and reduction of various pollutants. He finds that Weitzman's advice regarding the choice of quantity or price instrument does not hold in general, and the efficient choice of instrument for one pollutant will depend on the choice for the other pollutant.

Another strand of the literature looks at the potential for emissions intensity regulation to out perform fixed quantities or prices in the presence of uncertainty. Quirion (2005) finds that with uncertainty about business-as-usual emission levels and about the slope of the marginal cost curve, an absolute cap on emissions produces slightly higher expected welfare than a cap on emissions intensity, but a price instrument yields substantially higher expected welfare than an intensity cap. Pizer (2005) suggests that indexing emissions targets to a measure of economic growth is a good approach for dealing with economic growth and unexpected changes in economic fortunes. Pizer and Newell (2006) analyze the use of indexed regulation for climate policies and identify conditions (related to the first and second moments of the index and the ex post optimal quantity level of the emissions cap) under which indexing will improve welfare as compared to both fixed quantities and fixed emissions taxes.

A safety valve has obvious relevance with respect to the ability to respond to changes over time, and therefore has some relation to the opportunity to bank emission allowances. The relationship between emissions banking and a safety valve is little explored in the literature.

Jacoby and Ellerman (2004) suggest that banking will provide less protection from upside cost shocks than would a safety valve, particularly during the early years of a policy when no bank has yet accumulated for firms to draw on (assuming borrowing from the future is prohibited). However, they also point out that banking can provide greater price support in the case of lower than average cost, because the safety valve proposals usually do not include a price floor on allowances. Banking provides a way to capitalize on a short run decline in marginal abatement cost by enabling extra emission reductions in that period that can be banked for use in later periods when costs may be higher. However, if the decline in cost is long-term in nature then the price will fall in every period and banking will provide little price support.

3. The Single-Sided Safety Valve

The flexibility given to individual firms that is inherent in a cap and trade program has as its *raison d'être* the underlying variability in costs of the environmental policy at individual facilities. To other firms and the government, the variability in costs appears like uncertainty. Investors would be expected to take into account the distribution of potential outcomes when, so the underlying variability along with uncertainty about various factor prices and technical issues may be thought of as a fundamental characteristic of the problem.

We can begin to see the incentive effects of a policy feature such as a safety valve on investment behavior by considering the case of simple asymmetric information, wherein the investor has perfect foresight but the regulator has to make a decision in the absence of that information, perhaps before the information is revealed. For example, Title IV of the Clean Air Act Amendments regulating sulfur dioxide (SO_2) passed Congress in 1990 but the first phase of the program did not take effect until 1995, and the second took effect in 2000. In 1990 the delivered price of natural gas for electric utilities was about $3.15/kcf but by 1999 it had fallen to $2.89/kcf (2004$). Similarly, the average price for low sulfur subbituminous coal fell from about $12.81/ton in 1990 to $7.56/ton in 1999, while the consumption of low sulfur coal grew tremendously (EIA 2005, Tables 6.8 and 7.8). Investment decisions to comply with the legislation crafted by Congress in 1990 continued to take shape more than a decade later.

The safety valve has been suggested as a mechanism to insure against outcomes that differ widely from anticipated costs under a cap and trade program. For example, if price of an important factor of production were to rise higher than expected, potentially leading to unexpected costs of pollution control, the safety valve mechanism could issue additional emission allowances at a specified price thus effectively assuring that the marginal cost of pollution control could not rise above that price.

To illustrate the safety valve we conjecture a potential carbon dioxide (CO_2) emission reduction policy. We conjecture marginal benefits of emission reductions to be a known parameter and assume the regulator sets a target where marginal benefits equal expected but uncertain marginal cost. We evaluate the important case of natural gas price uncertainty using RFF's detailed simulation model of the electricity sector. The model divides the nation into 20 regions, 9 of which are assumed to yield electricity prices based on market prices, and the rest are assumed to be under cost of service regulation. The model includes the Clean Air Interstate Rule (CAIR) and the Clean Air Mercury Rule (CAMR) policies for NO_x, SO_2, and Hg emissions. We assume a discount rate of 8%, with 2030 as the forecast horizon year.[1]

The results for this example are illustrated in Figure A, where the horizontal axis represents the aggregate emissions of CO_2 from the electricity sector in 2020. We characterize the mid value for future natural gas prices to be $6.31/mmBtu under the policy. The central point in the figure is a quantity target (E^*) of 2,423 million tons where, under the mid value for natural gas price, we expect a marginal cost (P^*) of $31.15 per ton. We assume this is exactly equal to the marginal benefit, which is a known parameter. Under the assumption that marginal benefits are constant, the same outcome with respect to aggregate emissions could be achieved by setting an emission fee equal to P^*, which would result in an emissions level E^*.

<Insert **Figure A**. Illustration of the safety valve.>

The downward sloped line that lies above the expected permit price in Figure A illustrates a realization for natural gas prices in which prices are higher than expected, and equal to $10.16/mmBtu. This leads the cost of achieving emission reductions to be higher than in the mid case because shifting from coal-fired to gas-fired generation is an important way that emission reductions are achieved. For the high gas price case the marginal cost of achieving the emission target E^* increases to P^h = $50.55/ton. Since marginal benefits are assumed constant and equal to $P^* < P^h$, there is a welfare loss equal to the large shaded triangle because the quantity of emissions are too low given the high cost of emission reductions in the high gas case.

[1] Further detail on the model can be found in Paul and Burtraw (2002).

This example illustrates the way that a single-sided safety valve can improve welfare. Were there a safety valve in place, say at a level equal to the average of the mid and high allowance price outcomes, e.g. P^{HSV} = \$ 41/ton, it would cap the level at which marginal costs could rise by issuing additional emission allowances, leading to emissions of 2,568 million tons. Compared to the target where marginal benefit equals marginal cost (E^*, P^*), there would still be a welfare loss at P^{HSV} indicated by the smaller cross-hatched triangle, but this welfare cost would be less than from the strict quantity instrument without the safety valve.

4. The Symmetric Safety Valve

Were the safety valve to apply only when marginal costs are higher than expected the emission target would not respond if natural gas price to turns out to be lower than expected. The lower line segment in Figure A illustrates this outcome with a natural gas price in 2020 equal to \$4.42/mmBtu, where the marginal cost of achieving the emission target E^* decreases to P^L = \$17.57/ton. The welfare consequences of the drop in natural gas price can be just as great as when gas price is higher than expected due to the difference between marginal benefits and marginal costs. Since marginal benefits are assumed constant and equal to P^* there is a welfare cost analogous to the large shaded triangle in the previous example, but in this case the loss is due to the fact that from an efficiency perspective the quantity of emissions is too high given the low cost of emission reductions.

A low-side safety valve could correct for the unexpected decline in compliance cost. Were there a safety valve level at a level that was the average of the low and mid allowance price outcomes, P^{LSV} = \$24.31/ton, it would cap the extent to which marginal costs could fall by reducing the number of allowances provided to the market in the current or future periods. A reduction in emissions to E^{LSV} = 2,257 million tons would be achieved. There would still be a welfare cost compared to the efficient outcome ex post, but the cost would be less than under the strict quantity instrument without the safety valve.

There has been little attention given to how a safety valve would function. In the case of a high-side safety valve, advocates have suggested that the regulator could issue additional allowances at the safety valve price level through direct sale, or potentially through free allocation. The low-side safety valve has the same structure. If allowances fall were to the level of the floor, the regulator would reduce the provision of allowances in future periods. An example of this approach is embodied in the Clean Air Interstate Rule (CAIR) where the allowable quantity of future emissions changes the value (ton of emissions per allowance) of future allowances without changing the quantity of the allowances. If the program includes inter-period banking, this reduction in the number of allowances issued in the future will lead to an increase in the price of allowances in future and in the current period, as occurred with the promulgation of CAIR.

An even more direct way to implement the low-side safety valve would be through adjustments in an auction, were an auction to be used for to initially distribute a portion of the allowances. In this case the safety valve would directly resemble a reservation price in the auction, which is a common feature in auctions including previous auctions for the distribution of emission allowances. When the safety valve policy combines a high-side and a low-side safety valve, we refer to it as a symmetric safety valve.

5. Historical Experience

Historically, the failure to have a safety valve on the low side in the event that compliance costs are lower than expected has had larger consequences than the failure of a safety valve on the high side. The only important example of unexpected outcomes within a cap and trade program that may have been remedied by a safety valve on the high side has been the RECLAIM program in southern California, where prices skyrocketed in 2000 due to unexpected demand for emission allowances reflecting very high marginal cost of compliance, which led to suspension of trading in the program in 2001. In that program, however, emission allowance banking also could have helped remedy the market disruption.

In contrast, the most prominent economic failure of any cap and trade program has occurred in the SO_2 program under Title IV – a program generally noted for its many successful aspects. The SO_2 program is credited with success in facilitating the reduction in compliance costs compared to prescriptive regulatory approaches (Carlson et al. 2000; Ellerman et al. 2000), demonstrating on a large scale the effectiveness of an economic approach to pollution control (Stavins 1998; Joskow, Schmalensee and Bailey, 1998), and achieving billions of dollars in environmental and public health benefits.

However, the expensive failing of the SO_2 program has been its inability to adjust to new information. In 1990, at the adoption of Title IV, Portney (1990), the only economist who ventured an opinion about the benefits and costs of the amendments, concluded that the benefits of Title IV about equaled the cost. By the first year of the program’s implementation in 1995, it had become clear that the benefits would be an order of magnitude greater than costs (Burtraw et al. 1998). Unfortunately the program was unable to adapt to this new information until the adoption of CAIR, now scheduled to take effect fifteen years after the launch of the program.

Why did the estimates of benefits and costs change so dramatically? First, the anticipated benefits of emission reductions grew tremendously with new information about the damage to human health from fine particulates associated with emissions of SO_2 and NO_x. Second, and more important to this discussion, the estimates of the costs of emission reductions fell sharply, due in large part to the flexibility in compliance options afforded by the program.

The fact that information can change so dramatically and so quickly leads one to ask: To what extent does policy reflect scientific information about both the benefits and costs of regulation? Scientific and economic information is fundamentally uncertain. How policy-making interprets the data, and how the policy system responds when scientific information evolves, is of vital importance. Typically, once regulators reach a decision, it becomes exceedingly difficult to modify that decision (Center for International Studies, 1998). For instance, the Clean Air Act was amended in 1977, again in 1990, and has not been amended further since. Statutory regulation such as Title IV put regulators feet into cement. It is very difficult to change statutory direction given new scientific information.

The policy system could benefit from the use of decision rules that automatically incorporate new information. It is understandable that the policy system would be slow to incorporate new information about the benefits of Title IV, because information about benefits is not readily observable outside of the process of scientific research and peer review, which may take years to achieve general acceptance. However, cap and trade programs are uniquely designed to generate information about costs, in the form of allowance prices, which instantaneously provide a summary statistic of pollution control costs that is widely accessible.

<Insert **Figure B**. Variation in SO_2 prices.>

Before passage of CAIR, which directly influences compliance with the SO_2 trading program, estimates suggested that the expected SO_2 emissions in 2010 were to be about 9.18 million tons (Banzhaf et al. 2004). In 1990 the EPA estimated that the marginal cost of achieving the emission reduction targets in Phase II around the year 2010 would be $718-942/ton (2004$)(ICF 1990). However, as the program unfolded it quickly became apparent that the marginal costs as reflected in the price of emission allowances were dramatically below expectations. Figure B illustrates that the price has been well below $200/ton throughout most of years of the program.

Let us imagine that it was Congressional intent to roughly balance marginal benefits with marginal costs, and that a low-side safety valve had been in place that would reduce the provision of allowances were price to fall below $567/ton, about 33% below the mid-value of the range of expected costs. Banzhaf et al. estimate that an SO_2 allowance price of $567/ton in 2010 would yield total national annual emissions of 7.1 million tons, about 2.08 million tons less than under Title IV in the baseline (and in the absence of CAIR).

What would have been the value of a low-side safety valve that led to additional emission reductions? Banzhaf et al. use estimates of marginal benefits of $3,968/ton. This is substantially less than those used by the EPA in Regulatory Impact Assessment because Banzhaf et al. use a lower value of statistical life. Using the Banzhaf et al. estimates, the additional annual health benefits from placing a floor on the allowance price would total $8.25 billion in 2010 (2004$). Perhaps Congress could not have expected benefits of this magnitude from a safety valve, because it did not expect benefits to be this large. Alternatively, one could say that if Congress acted to equate marginal benefits and marginal costs then they would value additional emission reductions at an expected value of $718-942 per ton. At this value, the anticipated additional health benefits from the safety valve set 33% below expected marginal cost would be between $1.5 billion to $1.95 billion in 2010. Arguably, the legislative intent of Congress was to capture these benefits, but they did not have the policy tools available at the time to anticipate and flexibly adjust to changes in scientific information. The symmetric safety valve provides such a tool.

6. The Safety Valve Affects Expectations

The model illustrated in Figure A has a fundamentally naïve characterization of behavior because, as illustrated, the regulator makes decisions on the basis of expected values. She does not account for the effect of the safety valve affects expected values. Consequently the imposition of a one-sided safety valve will influence the market equilibrium and affect the decisions of investors, with unintended and potentially negative consequences that could undermine policy goals.[2].

[2] By analogy, the provision of insurance affects the behavior of investors because the insurance changes the expectations over potential pay-offs. Here we find something similar – investors can be expected to respond to the safety valve, which leads to a different market equilibrium.

We simplify the multi-period problem into an instantaneous present value calculation. Considering the profit function for a single firm that offers nonemitting electricity generation:

$$\pi = q \bullet P\left(Q, P_A\right) - C\left(q\right) \tag{1}$$

where q is the quantity produced by the potential investment, Q is the aggregate quantity in the market and P_A is price of allowances. Cost is a function of quantity of the production. The price of emission allowances is not included because the facility is nonemitting. We assume that the electricity price function and the cost function are increasing in their arguments.

The firm maximizes profits by choosing quantity (q). Under the assumption that the facility's output is too small to make an impact on the aggregate production and price, then $\frac{\delta P}{\delta q} = 0$ and the firm maximizes profits by choosing q such that marginal revenues equal marginal costs:

$$P\left(Q, P_A\right) = \frac{\delta C}{\delta q} \tag{2}$$

In general we expect the aggregate quantity and price of allowances to be uncertain, so that $Q = \tilde{Q}$ and $P_A = \tilde{P}_A$ (where the tilde represents uncertain variables), which cause the product price to be uncertain ($P = \tilde{P}$). Assuming the firm is risk neutral, the profit maximization condition would require the firm to equate expected marginal revenue with marginal cost: $\mathrm{E}\left(P\right) = \frac{\delta C}{\delta q}$. We will note the potential distribution f of allowance prices stretching from zero, bounded at the minimum, to infinity $\tilde{P}_A \sim f\left(0, \infty\right)$, which along with the distribution of potential aggregate generation determine the expected electricity price.

The high-side safety valve intentionally alters the distribution of the potential allowance price, so that the price cannot rise above the safety valve level (SV). If we naively ignore the interaction of the allowance price and the investment decisions of other firms and consider only the role of the safety valve on allowance price when the decisions of other firms are held constant, e.g. $\left(\frac{\delta Q}{\delta P_A}\right) \cong 0$, then the allowance price with the safety valve has the distribution:

$$\begin{aligned} \tilde{P}_A^{SV} &\sim h\left(0, SV\right) \\ &= f \quad \text{for } P_A \leq SV \text{ and} \\ &\quad SV \text{ for } P_A > SV. \end{aligned} \tag{3}$$

Letting F and H be the cumulative distribution functions for $\tilde{P}_A^{SV}$ and $\tilde{P}_A$, then $F \le H$ over their entire range, and $\tilde{P}_A^{SV}$ will have an expected value that is strictly less than P_A, $\mathrm{E}\left(P_A^{SV}\right) < \mathrm{E}\left(P_A\right)$.

This naïve characterization of the change in the distribution of potential allowance prices is illustrated in Figure C. The top panel illustrates a probability distribution for allowance price with the dotted curve. Allowance price is designated simply by “P” in the absence of a safety valve. The expected value for the allowance price is designated E(P), shown by a dotted line. The addition of the safety valve censors the potential distribution of allowance prices. If the distribution is otherwise unaffected, as described in equation (3), the mean shifts to the left, as indicated by the dashed line $E[Psv]_{\text{naïve}}$.

<Insert **Figure C.** Illustration of the distribution of allowance prices associated with uncertain gas price outcomes.>

A consequence of the change in the allowance price would be a change in the equilibrium in the electricity market, leading to a lower price under the safety valve, $\mathrm{E}\left(P^{SV}\right) < \mathrm{E}\left(P\right)$. The individual profit maximizing investor described in equation (1) would choose a level of production under the safety valve where:

$$\mathrm{E}\left(P^{SV}\right) = \frac{\delta C}{\delta q^{SV}} < \frac{\delta C}{\delta q} = \mathrm{E}\left(P\right) \tag{4}$$

leading to a reduction in its investment and output, $q^{SV} < q$.

The consequence of the high-side safety valve in this example is to reduce investment in the nonemitting facility. One can conjecture that in the aggregate the policy leads to less investment in renewable technology or low-emitting technology that may suffer a price disadvantage when the external social costs of electricity generation are not included in electricity price. The cap and trade program serves as a mechanism to internalize into investment decisions the social cost of technology choices and “level the playing field,” as many observers have suggested. However, the single-sided safety valve would appear to provide an asymmetric influence that would tilt the playing field away from investments in nonemitting sources.

7. The Safety Valve Equilibrium in a Model with Perfect Foresight

The formulation above assumes that the behavior of other investors or actors in the market does not respond to the change in expectations, and that aggregate quantity and characteristic of generation by other parties is unchanged due to the change in allowance price $\left(\frac{\delta\tilde{Q}}{\delta P_A}=0\right)$. However, clearly there would be a response. For instance, one could imagine that a lower allowance price would lead to more fossil generation, which would seem to lower electricity price and reinforce the effect described above. However, the lower allowance price also might increase the emission intensity of generation for any given level of production, which would cause a sort of bounce back in the price of allowances. In addition, whatever the underlying source of uncertainty for allowance price is, it is also likely to affect directly the cost and aggregate quantity of production.

In Figure C the probability density function for potential allowance price in the new equilibrium under the safety valve is illustrated by the solid curve in the top panel, and the solid line illustrates the new expected value of the distribution. The bottom panel illustrates the shift in the cumulative distribution function. We conjecture that the new equilibrium yields an expected value that is between the other two measures that are illustrated. But the equilibrium outcome in a general model is difficult to anticipate without simulation modeling. Therefore we return to that platform. In doing so, we conjecture *a priori* that the high-side safety valve should lead to less investment in nonemitting and low-emitting sources of generation than in the absence of the safety valve, as well as a lower expected allowance price, a lower electricity price and greater expected emissions.

In the simulation we explore underlying uncertainty about natural gas prices in the future. As the central case we adopt EIA (2006) forecasts reported in the *Annual Energy Outlook.* We consider two alternatives, which are labeled high and low gas price cases and incorporate a 30% increase and decrease in gas prices. We assume gas price is normally distributed. High and low prices are picked to represent prices that are one standard deviation away from the mean.

In this modeling exercise we freeze natural gas and coal prices at the assumed forecast values in each year and thus these fuel prices are not allowed to vary with the level of fuel used. We also freeze the level of electricity consumption in order to avoid second best issues in the welfare calculation that are associated with differences between price and marginal cost. For all of the pollution policies emission allowances are allocated to emitters on the basis of historic generation and additional permits are purchased at the safety valve price.

<Insert **Table 1**. Deterministic model with certain foresight.>

The equilibria that are achieved under each scenario are summarized in Table 1. In each case, the model is deterministic and actors behave as though they have certain and perfect foresight – e.g. they know the future path of natural gas prices and respond accordingly. The middle column represents the mid case for gas prices. The first and last columns represent the outcome for low and high gas prices respectively, in the absence of a safety valve. There is little change in CO_2 emissions, but it is interesting to note that low gas prices lead to a modest increase in emissions because there is new gas generation in lieu of new investment in renewables. High gas prices also lead to an increase in emissions, as gas-fired generation falls and there is an increase in coal-fired generation that more than offsets the new investment in renewables. Allowance price ranges widely from a low of $33 under the low gas scenario to a high of $74.Electricity price also ranges widely. Figure D illustrates the change in electricity price relative to the mid case for each simulation year in the model.

<Insert **Figure D**. Variation in electricity prices in the deterministic model with perfect foresight.>

The second column of Table 1 represents the influence of a safety valve on the low side, which we label a symmetric safety valve. The fourth column represents a single-sided safety valve on the high side. On either side, the safety valve has a direct effect on CO_2 emissions, as would be expected because it affects the quantity of emissions directly. As a consequence the variation in other variables such as electricity price and renewable generation is reduced, compared to the absence of the safety valve.

Note also that in either case, the safety valve improves welfare relative. Welfare is calculated as the sum of changes in producer and consumer surplus, plus the change in environmental benefits associated with changes in emissions relative to the emission quantity target, valued at their expected cost of $51 /ton. The change in welfare for each case is measured relative to the mid gas case. The greatest improvement comes from adding a safety valve in the high gas price case. Relative to the mid gas price case, which is normalized to a value of zero, the high gas price case leads to a loss of over $23 billion. The high-side safety valve reduces this loss to about \$7 billion because it closes the gap between marginal benefits and marginal costs by allowing an increase in emissions. In the low gas case, welfare improves by over \$37 billion, due to lower cost of production. In the low-side safety valve case, welfare improves further to \$40 billion by reducing emissions below the emission target, thereby taking advantage of the relatively low marginal cost of abatement.

8. Modeling Uncertainty

The simulation model is deterministic, meaning that it incorporates certain foresight about potentially uncertain variables. Investment decisions are made as though each actor knows for certain the future values of every variable, as well as the decisions of every other actor, so there is no uncertainty taken into account in the model solution. However, although the model is itself deterministic, we can use a collection of model solutions to make a mathematical inference about the outcome of the market equilibrium when investors make decisions taking uncertainty into account.

Using the results from the deterministic model for various realizations of the underlying uncertain parameter, we construct a linearization using the delta approach, which is a variation of a Taylor series expansion. The expected value of a function ϕ of a random variable $\tilde{g}$ with expected value $\bar{g}$ and variance σ_g^2, can be approximated by:

$$\begin{aligned} \mathrm{E}\left[\phi(\tilde{g})\right] &\cong \phi(\bar{g}) + \phi'(\bar{g})\mathrm{E}\left[(\tilde{g}-\bar{g})\right] + \frac{1}{2}\phi''(\bar{g})\mathrm{E}\left[(\tilde{g}-\bar{g})^2\right] \\ &= \phi(\bar{g}) + \frac{1}{2}\phi''(\bar{g})\sigma_g^2 \end{aligned} \quad (5)$$

where ϕ' and ϕ'' ' are first and second derivatives of the function.

The function ϕ can represent a variety of measures that we are interested in including aggregate economic welfare, electricity price, allowance price or the installed nonemitting generation capability. For this experiment, the random variable $\tilde{g}$ is the natural gas price. We consider low, mid and high values of \$4.42/mmBtu, \$6.31/mmBtu, and \$8.21/mmBtu in 2020 (2004\$). We assume it is common knowledge that these prices are distributed normally with an expected value of \$6.31/MMBtu and a standard deviation of \$1.90/MMBtu, so the mid value in this experiment is the mean value of the natural gas price and the low and high values are both one standard deviation from the mean.

<Insert **Table 2.** Delta method approximation of key variables in model with uncertainty.>

The results from this experiment are reported in Table 2, for the case of no safety valve, a high-side (only) safety valve, and a symmetric safety valve. The high-side safety valve leads to the expectation of greater emissions than in the no safety valve case because with some probability the safety valve will be triggered, thereby placing extra allowances on the market. As a consequence the allowance price and electricity price are lower. All variables except welfare are normalized using the no safety valve case as a numeraire (the value is set equal to one). For welfare, the difference between the no safety valve case and the mid case in the deterministic model is normalized as a numeraire because only changes in welfare have economic relevance. A potentially important unintentional result is that the lower expected allowance price leads to lower expected payoffs to investment in renewable technologies. Consequently we see a decline in renewable generation. Here, only a subset of renewable technologies is allowed to change because biomass is held constant. Were biomass also allowed to change one would see even more of an effect on renewable generation.

Many observers have criticized the high-side safety valve because it might undermine the environmental targets of the program, and that is the result we obtain. Emissions are higher and investments in new technology are lower as a result of the safety valve. The reduction in investments initiates a cascade of consequences, as there is less learning as a result of the decline in investment, so the costs of renewable technologies remain above their levels in the absence of the safety valve.

However, the unintended consequences are fully remedied when the safety valve is characterized as a symmetric instrument. In this case, emissions fall back to virtually the same level as in the absence of a safety valve, and renewable investments increase to above their level in the absence of a safety valve. The results for the high-side and symmetric safety valve are compared visually in Figure E. The figure shows that not only do measures of interest to environmental advocates return to their intended levels, but welfare improves even further than in the case with only a high-side safety valve. Also, electricity price and allowance price return to nearly the same level as in the absence of the safety valve.

<Insert **Figure E.** Delta method approximations of outcomes under uncertainty.>

9. Surprise in a Model with Certain but Imperfect Foresight

The delta method could be applied in a different way by assuming a different information structure. In the previous example, the finite differences are calculated using the model with perfect foresight. An alternative would be certain but imperfect foresight; wherein investment decisions made under one set of assumptions could prove imprudent were conditions to change unexpectedly. For example, if gas prices deviate from expectations after investment decisions have been made, then generators could experience large losses in profits and welfare could be negatively affected. Since the safety valve is a policy attempt to mitigate the welfare costs of surprises such as this one, we consider a case where investors' expectations are incorrect, and use these data to calculate finite differences.

The scenario involves a surprise in natural gas prices in 2015. Investors make an investment plan based beginning in the first simulation year in 2010 and based on certain but imperfect foresight about the future path of gas prices. In 2015, investors learn that gas prices are on a different path. Taking existing investments as sunk, investors solve the perfect foresight with the new data. We ran simulation scenarios that include a gas price surprise to determine the effect both a one-sided and symmetric safety valve would have on the expected value of several key variables.

Figure F illustrates the path of electricity prices under the surprise in natural gas prices, compared against the expected price path for prices that was illustrated previously in Figure D. The surprise in 2015 leads to a precipitous change in electricity prices in the absence of a safety valve, especially when natural gas prices rise unexpectedly.

<Insert **Figure F.** Variation in electricity prices in the model with certain but imperfect foresight.>

The surprise in gas prices lead to comparable variations across the different policy scenarios than were obtained in the previous example for most variables. Table 3 illustrates these differences. One outcome that is interesting is the increase in renewable generation in the high gas price case with a high-side safety valve. The reason is that although dedicated biomass does not change in the model, co-fired biomass is allowed to change. The high gas price leads to more coal-fired generation, and with that comes a greater amount of co-fired biomass.

<Insert **Table 3.** Model with certain but imperfect foresight and a gas price surprise in 2015. Results for 2020.>

We apply to delta method to this set of results to replicate the experiment of a first-order approximation to behavior in a model with uncertainty. Table 4 reports these results, and they are illustrated visually in Figure G. Again, the variables of interest return to their approximate levels in the absence of the safety valve. The effect on renewable generation is greater than in the previous example. Also, the welfare contribution of a symmetric safety valve is greater relative to the high-side safety valve.

<Insert **Table 4.** Delta method approximation of key variables in model with certain but imperfect foresight, results for 2020.>

<Insert **Figure G.** Delta method approximations of outcomes in a model with certain but imperfect foresight.

10. Conclusion

Significant attention has been directed to price stabilization measures in emission allowance trading programs. In particular, attention has focused on the introduction of a single-sided safety valve that would mitigate potential price spikes by introducing additional emission allowances into the market when costs rise above the specified “safety valve” level. However, experience with such programs indicates that the most important examples of price volatility to date have occurred when allowance prices fell below their expected values. For example, in the case of SO_2 emission trading, the inability of the trading program to adjust to the fall in allowance prices led to welfare losses of between $1.5 and $8 billion dollars per year.

A second reason to be interested in a price stabilization mechanism when prices fall below expectations is the influence that low prices have on investment. In the absence of a safety valve, investors will take risks given expectations over a distribution of potential payoffs for their investment. A high-side safety valve that prevents spikes in allowance prices will have the unintended consequence of lowering the overall expected allowance price, and as a consequence the overall expected return on an investment in nonemitting technology.

A symmetric safety valve solves both these problems. A symmetric safety valve is a price stabilization policy that works in the case of unanticipated spikes or drops in allowance price. In the case when allowance price falls below the safety valve floor, the safety valve would contract the number of allowances issued in the market. The reduction in the quantity of allowances can be implemented in a variety of ways, but the simplest way may be through a change in the portion of emission allowances that is initially distributed through auction. In fact, good design suggests that an auction should have a reservation price, which is a floor below which the allowances will not be sold. Such a price floor serves directly to implement the low-side safety valve.

We use a linear approximation representing a Taylor series expansion around the mid case to model uncertain natural gas prices in a detailed electricity market model. We show that a high-side safety valve can be expected to increase emissions and decrease investment in nonemitting technologies, relative to the absence of a safety valve. However, the symmetric safety valve returns the expected value for these and other key parameters to the vicinity of their levels in the absence of a safety valve. In addition, although a high-side safety valve improves welfare, a symmetric safety valve improves welfare even further. In summary, we find a symmetric safety valve can improve the performance of allowance trading programs, improve welfare, and may help overcome political objections from environmental advocates who have opposed the use of a safety valve.

Two areas remain to be developed in this analysis. One has to do with a method for determining the breadth of a safety valve around expected marginal costs. When marginal benefits are constant, as in the examples we use, then the most efficient safety valve would be one exactly equal to the value of marginal benefits. In other words, the efficient policy is a tax. However, when marginal benefits are not flat but vary over a range then intuition suggests the efficient safety valve would vary from the expected level of marginal benefits.

A second area to be developed is the relationship between the idea of a safety valve and other approaches to stabilizing prices such as the so-called “circuit breaker.” The circuit breaker approach would anticipate a tightening of an emission cap over time, but modify that path in response to fluctuations in prices. If the price rose above a specified level, the reduction in the emission cap would be delayed. This approach has a strong similarity to a safety valve. In work currently in progress we are developing the analytical similarities of these two approaches.

References

Baumol, William J. and Wallace E. Oates. 1988. *The Theory of Environmental Policy,* Cambridge University Press (second edition).

Banzhaf, H. Spencer, Dallas Burtraw and Karen Palmer. 2004. “Efficient Emission Fees in the US Electricity Sector,” *Resource and Energy Economics*, 26: 317-341.

Burtraw Dallas, and Karen Palmer 2006. “Summary of the Workshop to Support Implementing the Minimum 25 Percent Public Benefit Allocation in the Regional Greenhouse Gas Initiative,” Resources for the Future Discussion Paper 06-45.

Burtraw, Dallas, Alan J. Krupnick, Erin Mansur, David Austin and Deirdre Farrell. 1998. “The Costs and Benefits of Reducing Air Pollutants Related to Acid Rain,” *Contemporary Economic Policy*, Vol. 16 (October): 379-400.

Carlson, Curtis, Dallas Burtraw, Maureen Cropper and Karen Palmer. 2000. “SO_2 Control by Electric Utilities: What are the Gains from Trade?” *Journal of Political Economy* 108: 1292-1326.

Center for International Studies, 1998. “Environmental Policymaking: A Workshop on Scientific Credibility, Risk and Regulation, Massachusetts Institute of Technology, Cambridge, September 24-25. Summary prepared by Brian Zuckerman and Sanford Weiner.

Ellerman A. Denny, et al. 2000. *Markets for Clean Air*, Cambridge University Press.

Energy Information Administration, 2005. *Annual Energy Review 2005.* Report No. DOE/EIA-0384 (2005). Posted: July 27, 2006 <http://www.eia.doe.gov/emeu/aer/txt/ptb0608.html>.

———. 2006. *Annual Energy Outlook 2006 with Projections to 2030.* Report #: DOE/EIA-0383(2006) (February).

Environmental Protection Agency. 1990. “Comparison of the Economic Impacts of the Acid Rain Provisions of the Senate Bill (S.1630) and the House bill (S.1630).” Report prepared by ICF Resources Inc. Washington: Environmental Proetction Agency, July.

Evans, David, 2006. *Integrated Environmental Regulation with Multiple Pollutants and Uncertain Joint Abatement: Theory and an Application to Electric Utilities.* Ph.D. Dissertation (in preparation), Department of Economics, University of Maryland.

Harrington, Winston, Richard D. Morgenstern and Peter Nelson. 2000. On the Accuracy of Regulatory Cost Estimates. *Journal of Policy Analysis and Management* 19:2, 297-317.

ICF Resources Inc. 1990. *Comparison of the Economic Impacts of the Acid Rain Provisions of the Senate Bill (S.1630) and the House Bill (S.1630[sic]).* Draft report prepared for the U.S. Environmental Protection Agency, July.

Jacoby, Henry D. and A. Denny Ellerman. 2004. The Safety Valve and Climate Policy. *Energy Policy* 32(4):481-491.

Joskow, Paul L., Richard Schmalensee and Elizabeth M. Bailey. 1998. “The Market for Sulfur Dioxide Emissions.” *American Economic Review*, 88: 4 (September): 669-85.

Kopp, Raymond, Richard Morgenstern, Billy Pizer and Michael Toman, 2002. “A Proposal for Credible Early Action in U.S. Climate Policy,” Resources for the Future (available at www.weathervan.rff.org/features/feature060.html accessed 8/14/02).

Newell, Richard G. and William A. Pizer. 2006. Indexed Regulation, Resources for the Future Discussion Paper 06-32, June.

Paul, Anthony and Dallas Burtraw, 2002. *The RFF Haiku Electricity Market Model*, RFF Report (May 22).

Pizer, William A. 2002. Combining price and quantity controls to mitigate global climate change. *Journal of Public Economics* 85 (3):409-434.

———. 2005. The case for intensity targets. *Climate Policy* 5 (4):455-462.

Pizer and Newell 2006

Portney, Paul R. 1990. “Economics and the Clean Air Act,” *Journal of Economic Perspectives,* 4:4, 173-181.

Quirion, Philippe. 2005. Does uncertainty justify intensity emission caps?. *Resource and Energy Economics* 27:343-353.

Roberts, Marc J., and Michael Spence. 1976. Effluent Charges and Licenses Under Uncertainty. Journal of Public Economics 5 (3-4):193-208.

Stavins, Robert N. 1996. Correlated Uncertainty and Policy Instrument Choice. Journal of Environmental Economics and Management 30 (2):218-232.

Stavins, Robert N. 1998. “What Can We Learn from the Grand Policy Experiment? Lessons from SO_2 Allowance Trading,” *Journal of Economic Perspectives*, 12: 3 (Summer), 69-88.

Weitzman, Martin L. 1974. “Prices vs. Quantities.” *Review of Economic Studies* 41 (4):477-491.

Weitzman, Martin L. 1978 "Optimal Rewards for Economic Regulation," *American Economic Review*, 68: 4 (Sept.) pp. 683-691.

Wenders, J.T., 1975. “Methods of Pollution control and the Rate of Change in Pollution Abatement Technology,” *Water Resources Research II* (June), 343-6.

Figure A. Illustration of the safety valve.

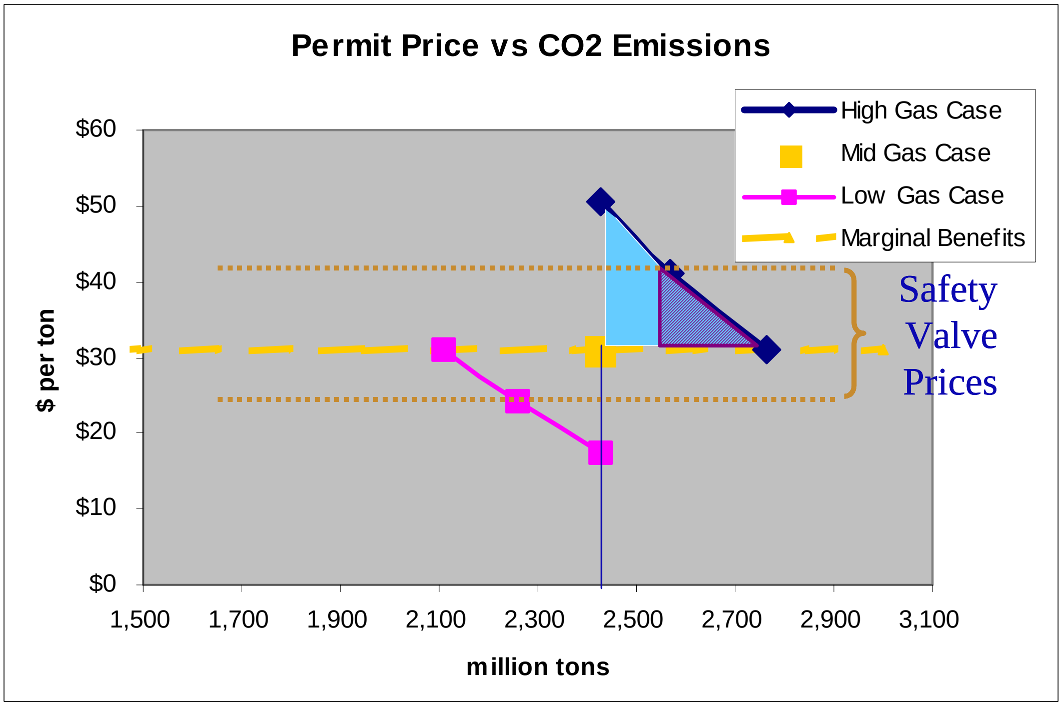

Figure B. Variation in SO_2 Prices

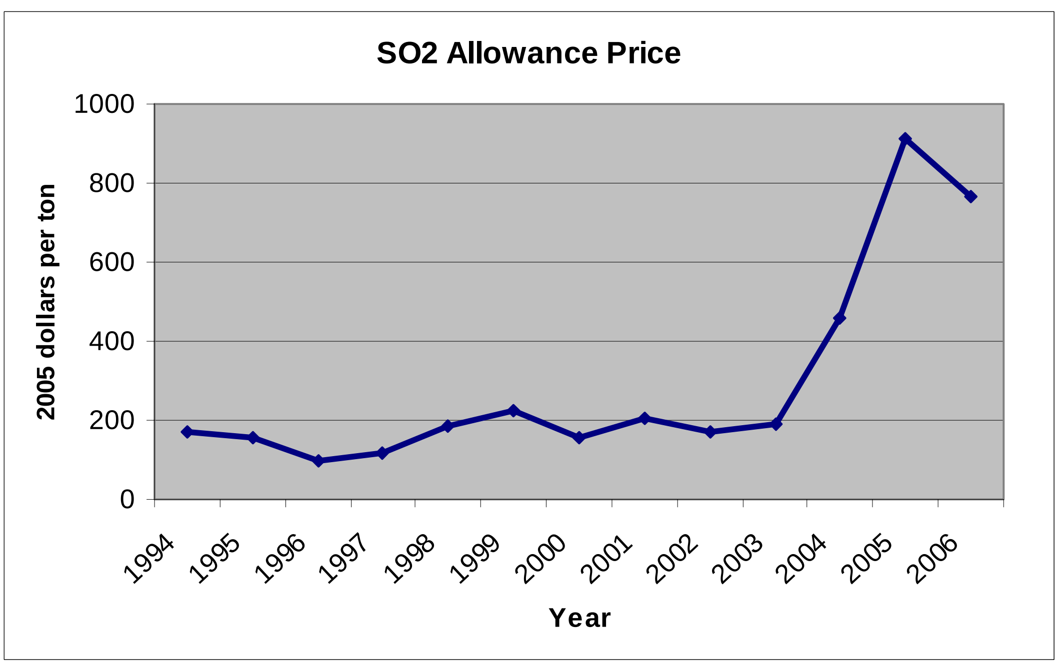

Figure C. Illustration of the distribution of allowance prices associated with uncertain gas price outcomes.

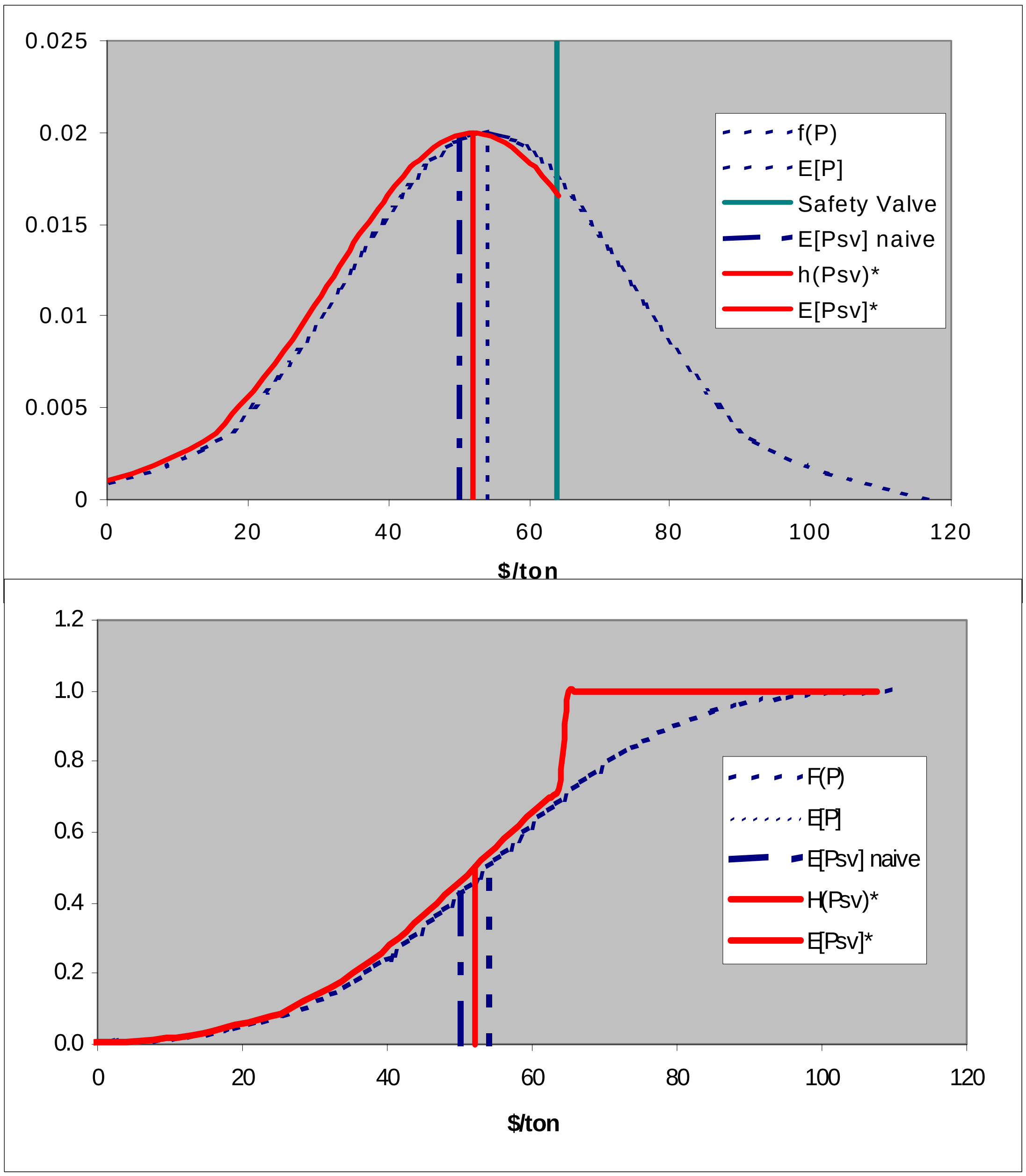

Figure D. Variation in electricity prices in the deterministic model with perfect foresight.

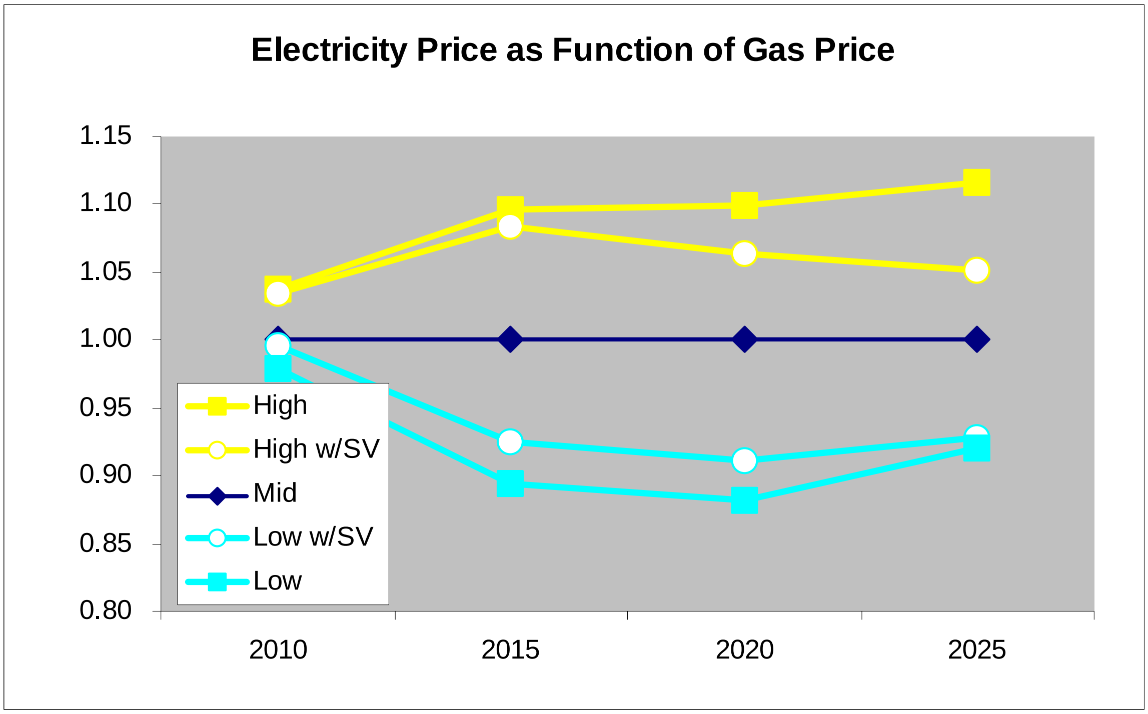

Figure E. Delta method approximations of outcomes under uncertainty.

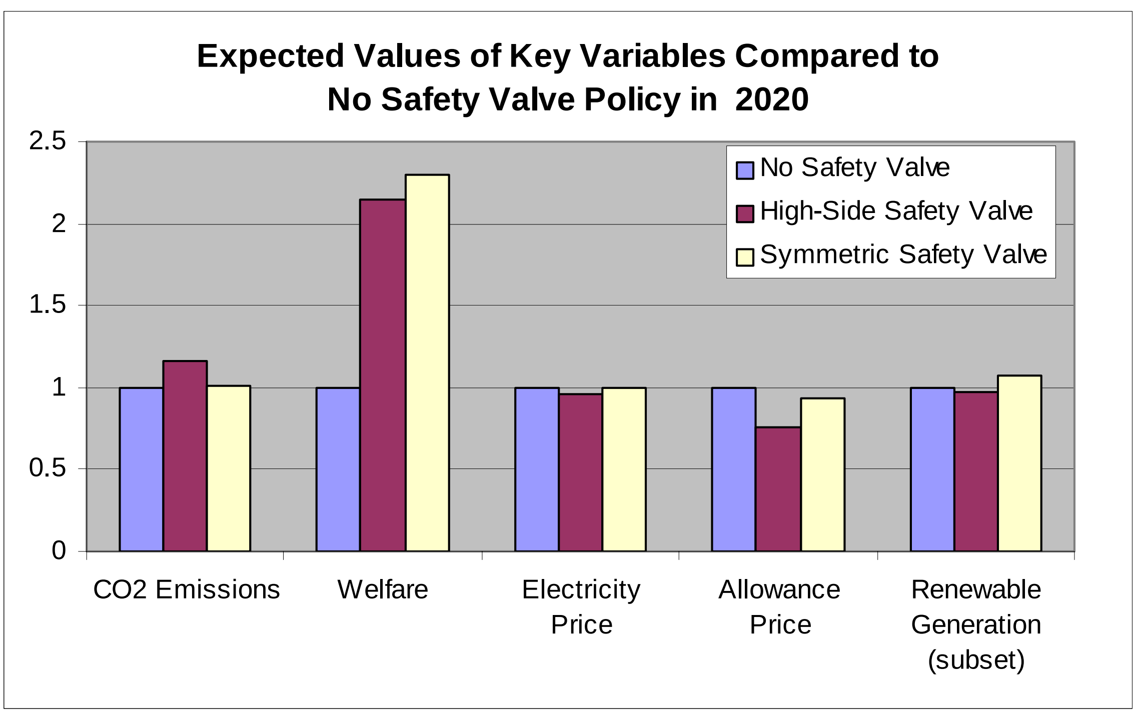

Figure F. Variation in electricity prices in the model with certain but imperfect foresight.

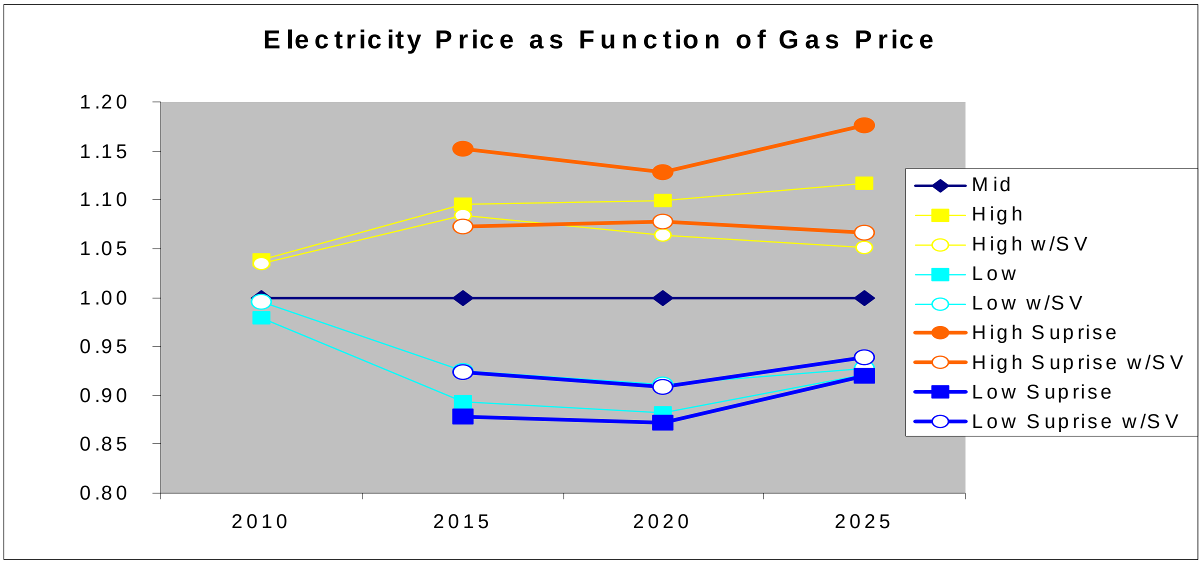

Figure G. Delta method approximations of outcomes in a model with certain but imperfect foresight.

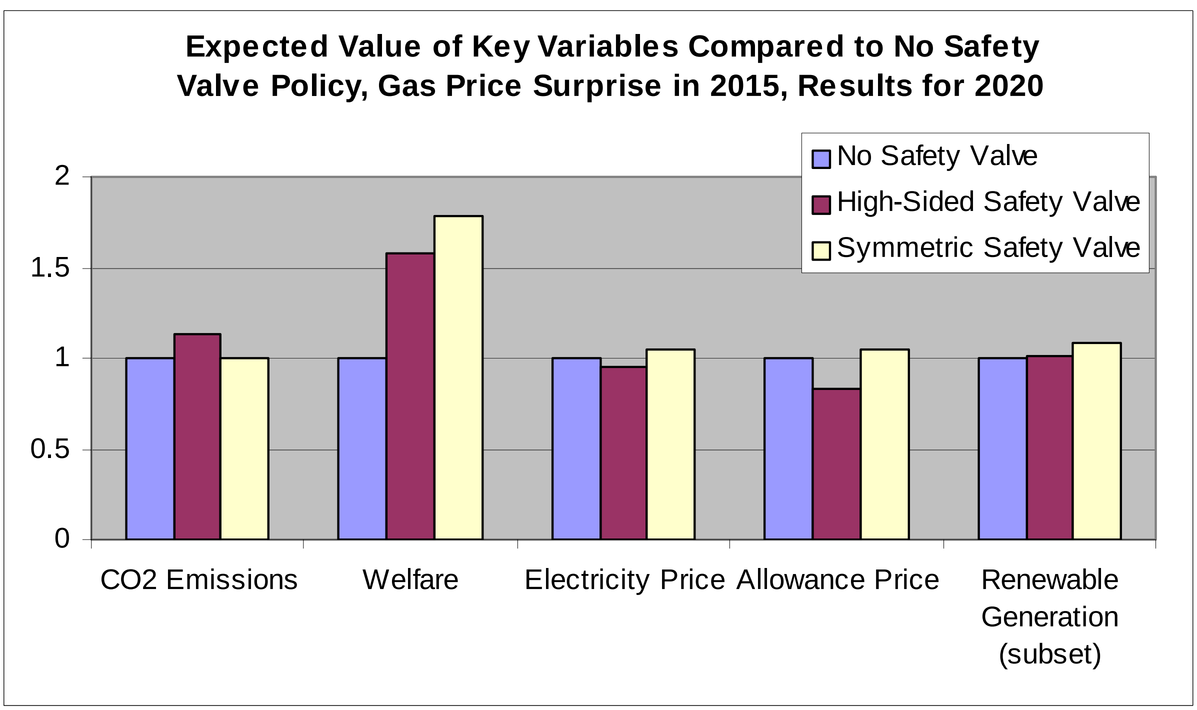

Table 1. Deterministic model with certain foresight, 2020.

	Low Gas Price	Low Gas Price w/ Symmetric Safety Valve	Mid Gas Price	High Gas Price w/ High-Side Safety Valve	High Gas Price
Gas Price ($/mmBtu)	4.42		6.31	8.21	
CO_2 Emissions (Mtons)	2,009	1,699	1,973	2,293	1,999
Welfare* (Billion $)	37.66	39.94	0	-6.85	-23.25
Electricity Price ($/MWh)	82.1	84.8	93.1	99.1	102.4
Allowance Price ($/ton)	33.0	43.5	51.1	59.7	74.1
Renewable Generation** (BkWh)	313	360	394	568	581

* Welfare compared to Mid Gas Price case.
** Underestimate because renewables include only changes in wind, landfill gas, geothermal. Biomass is held constant.

Table 2. Delta method approximation of key variables in model with uncertainty, 2020.

Expected Value $\mathrm{E}\left[\phi(\tilde{g})\right]$	**No Safety Valve**	**High-Side Safety Valve**	**Symmetric Safety Valve**
Gas Price ($/mmBtu)	6.31	6.31	6.31
CO_2 Emissions (Mtons)	1,983	2,313	2,015
Welfare* (Billion $)	13.82	29.62	31.80
Electricity Price ($/MWh)	91.43	88.19	90.78
Allowance Price ($/ton)	55.66	41.88	52.00
Renewable Generation** (BkWh)	494.5	482.3	528.2

* Welfare is the difference relative to the Mid Gas Price case.

**Underestimate because renewables include only changes in wind, landfill gas, geothermal. Biomass is held constant.

Table 3. Model with certain but imperfect foresight and a gas price surprise in 2015. Results for 2020.

	Low Gas Price	**Low Gas Price w/ Symmetric Safety Valve**	**Mid Gas Price**	**High Gas Price w/ High-Side Safety Valve**	**High Gas Price**
Gas Price ($/mmBtu)	4.42		6.31	8.21	
CO_2 Emissions (Mtons)	1,983	1,690	1,973	2,264	1,983
Welfare* (Billion $)	35.25	38.01	0	-13.85	-21.99
Electricity Price ($/MWh)	81.25	84.67	93.14	100.40	105.10
Allowance Price ($/ton)	32.14	43.28	51.14	59.42	68.42
Renewable Generation** (BkWh)	296	328	394	473	472

* Welfare is the difference relative to the Mid Gas Price case.

**Underestimate because renewables include only changes in wind, landfill gas, geothermal. Biomass is held constant.

Table 4. Delta method approximation of key variables in model with certain but imperfect foresight, results for 2020.

Expected Value $\mathrm{E}\left[\varphi(\tilde{g})\right]$	**No Safety Valve**	**High-Side Safety Valve**	**Symmetric Safety Valve**
Gas Price (\$/mmBtu)	6.31	6.31	6.31
CO_2 Emissions (Mtons)	1,992	2,256	1,982
Welfare* (Billion \$)	12.67	19.97	22.59
Electricity Price (\$/MWh)	93.18	88.84	92.04
Allowance Price (\$/ton)	49.47	41.15	51.57
Renewable Generation** (BkWh)	374.6	377.1	407.4

* Welfare is the difference relative to the Mid Gas Price case.

**Underestimate because renewables include only changes in wind, landfill gas, geothermal. Biomass is held constant.

Output-Based Allocation of Emissions Permits for Mitigating Tax and Trade Interactions*

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ABSTRACT

The allocation of tradable emissions permits has important efficiency as well as distributional effects when tax and trade distortions are taken into account. We compare different rules for allocating carbon allowances within sectors (lump-sum grandfathering, output-based allocation (OBA), auctioning) and among sectors (historical emissions or value-added shares). The output subsidies implicit in OBA mitigate tax interactions, unlike grandfathering. OBA with sectoral distributions based on value added is similar to revenue recycling with auctioning. OBA based on historical emissions supports heavier polluters, more effectively counteracting carbon leakage, but at higher welfare costs. Less energy-intensive sectors are also sensitive to allocation rules.

I. INTRODUCTION

Increasingly in recent years, many countries have been incorporating economic instruments into environmental policy, particularly "cap-and-trade" policies that fix emissions limits and allow firms to trade the rights to emit up to that cap. The United States is expanding its use of marketable emissions permits from sulfur dioxide (SO_2) to nitrogen oxides (NO_x) and potentially mercury, and proposals to reduce CO_2 include emissions trading. The European Union (E.U.) is proceeding with an emissions trading system for controlling greenhouse gases. Other countries, including Canada, are considering an emissions trading program for greenhouse gases.

When emissions are capped, they become a scarce and valuable resource. An important political—and economic—question is how to allocate these pollution rents. Most economists, citing the large literature on the "double dividend," recommend that permits be auctioned so that the revenues can be used to lower other distortionary taxes in the economy that otherwise increase the cost of environmental regulation. In practice, however, governments prefer to forgo the revenues and allocate the permits gratis to the industries covered by the trading system. For example, in its acid rain program, the United States distributed all of the SO_2 allowances, less a small reserve, to existing coal-fired power plants, an annual value in the range of $2 billion. The European Union has mandated that member states freely distribute their permits, imposing a maximum of 5% for auctioning permits in the first trading period (2005–2007). The McCain-Lieberman Climate Stewardship Act of 2003, the main proposal on the table to limit greenhouse gas emissions in the United States, provides for sector-based allocations but also some share (to be specified) to go a special nonprofit corporation that would be established to benefit consumers, the Climate Change Credit Corporation (CCCC).

A common method for gratis allocation is "grandfathering," which gives participating firms a fixed number of allowances. That number is often determined by historical emissions or market shares, but the key aspect is that the allocation does not vary with changes in circumstances. In the absence of other market failures, this lump-sum allocation offers the same incentives and efficiency as auctioning permits. However, in giving away all the permits, the value of the rents transferred to incumbent firms can vastly outweigh their actual cost burden from the regulation.[i] Any increase in marginal production costs tends to be passed on to consumers, who get no relief from the allocation. Another distributional concern is that firms that enter later may not get any allocation and have to purchase all their permits on the market. Thus, attention is turning toward allocation methods that can be updated (or will update themselves) according to changing market conditions and composition.

Accordingly, one method of updating that is frequently advanced is output-based allocation (OBA). For example, a cap may still be placed on the emissions of several sectors and each sector would be granted a fixed number of those permits, but *within* each sector, individual firms would receive a share of the sector cap in proportion to their share of their industry's output. As market shares within an industry change, permit allocations are updated, such that each unit of output is accorded the same average permit allocation. However, since output is a control variable of the firm, the allocation policy itself has behavioral effects, which in turn tend to reduce the efficiency of the environmental policy. Specifically, the allocation creates a subsidy to output, which effectively distributes the emissions rents to consumers in the form of lower prices, relative to lump-sum allocations. However, from an efficiency perspective, that subsidy also limits incentives to reduce emissions through conservation and diverts attention toward lowering energy intensity.

Environmental policy, of course, does not operate in a vacuum. The efficiency of a standard Pigouvian tax or an equivalent emissions permit system relies on the assumption that markets are not otherwise distorted. Where distortions exist, environmental policy may exacerbate them, rendering simple Pigouvian policies suboptimal. We focus on two major examples that can justify support for output: (1) emissions leakage and (2) tax interaction.

Emissions leakage occurs when significant portions of an emitting industry escape regulation. This problem is of particular relevance for transboundary pollutants, since foreign sectors will be outside the jurisdiction of a domestic regulator. In greenhouse gas policy, this issue is known as carbon leakage. Since they bear no environmental burden, excluded producers suddenly have relatively low costs compared with participants. Industry production then tends to shift away from participants toward nonparticipants (who are still emitting costlessly). An output subsidy for participants would discourage such intraindustry shifting of production and emissions. Bernard et al. (forthcoming) show that when the products of the exempt firms cannot be taxed to reflect the value of the embodied emissions, an output subsidy may be warranted—to the extent these products are close substitutes for those produced by regulated firms.

Taxing labor income creates another kind of imperfection by distorting the labor-leisure trade-off; in a sense, it taxes all consumption goods at the same rate, making them more expensive and making consuming leisure more attractive relative to consuming goods.[ii] Adding an environmental policy that makes some consumption goods even more expensive further distorts this trade-off. Environmental policies that raise revenues that can be used to lower distorting labor taxes unambiguously raise welfare from the no-policy scenario. Policies that do not raise revenue (like grandfathered permits) must have positive environmental benefits that outweigh the increased deadweight loss from the labor tax on the margin.[iii]

By providing a subsidy to output, output-based rebating may mitigate some of the impact of the tax interaction effect compared with lump-sum distribution. The implicit subsidy lowers the price of the dirty good, making goods consumption in general less expensive and real wages higher. However, the gain from a reduced disincentive must be balanced against the higher abatement cost of achieving the same level of emissions reduction. The net result may (or may not) be an improvement over distributed permits in this situation.

Goulder et al. (1999) show that performance standards can generate fewer efficiency costs than distributed permits in this second-best system. In their model, performance standards are less costly the less abatement is to be done by output adjustment than by emissions rate adjustment.[iv] On the other hand, Jensen and Rasmussen (2000), using a general equilibrium model of the Danish economy, find that allocating emissions permits according to output dampens sectoral adjustment but imposes greater welfare costs than grandfathered permits. Dissou (forthcoming), in an application to greenhouse gas reductions in Canada, finds that performance standards can mitigate losses in gross domestic product, but welfare is lower relative to grandfathered permits.

Although performance standards bear similarities to output-based allocation, there are some important differences. Performance standards (particularly tradable ones) allocate permits according to output and the target emissions intensity. In theory, they could be set so as to equate marginal abatement costs across multiple sectors.[v] Such a result can be replicated with a multisector cap-and-trade system with OBA, but with a single set of allocations at the sector level. In practice, many different sectoral allocation rules are possible—and more plausible—than expected equilibrium average emissions. Since these sectoral allocations determine the relative subsidy for output, we will closely examine the effects of different rules for dividing an emissions cap among sectors.

We look at how well OBA can address these second-best efficiency issues relative to grandfathering or auctioning emissions permits. We begin with some theoretical background, using a partial-equilibrium analytical model, before proceeding to the general-equilibrium numerical analysis. We use a version of the computable general equilibrium (CGE) model GTAP-EG, modified to incorporate a labor-leisure choice, to look at how well OBA can address issues of equity and efficiency relative to grandfathering emissions permits or auctioning with revenue recycling and with trade effects.

We find that the rules for setting the initial sectoral caps play an important role in determining the changes in welfare, industrial production, employment, and trade induced by the emissions policy. OBA with sectoral distributions based on value added generates effective subsidies more like a broad-based tax reduction, performing nearly like auctions and clearly outperforming lump-sum allocations. OBA based on historical emissions supports the output of more polluting industries, which more effectively counteracts carbon leakage, but is more costly in welfare terms. With less contraction among polluting sectors, more reductions must be sought among less carbon-intensive sectors and final demand, signaled by a higher carbon permit price. However, due to the importance of the tax interaction problem, historical OBA remains less costly in net welfare terms than traditional permit grandfathering, at least for targets that are not too stringent. In all cases output-based rebating is less efficient than auctioned permits, which raise revenues that offset labor taxes, encourage more work, and achieve this in a manner that does not distort the relative prices of dirty and clean goods.

II. PARTIAL EQUILIBRIUM MODEL

To build intuition for the first-order effects of different allocation mechanisms, we first present a partial equilibrium model of the affected industry. In the next section, we incorporate these industry incentives into a general equilibrium setting, to better account for the incidence of allocation on all prices and interactions with the broader tax system.

Consider a perfectly competitive industry with a representative firm that is a price taker in both product and emissions markets. The unit cost function, $c(.)$, is represented as a decreasing, convex function of the emissions rate μ. In other words, the firm chooses a technological or input mix that implies a given emissions rate, and exhibits constant returns to scale, which corresponds to a constant per-unit cost. Let y be the output of our representative firm. Let p be the market price for the good produced, while the price of a pollution permit is measured by t. Consumer inverse demand $p(y)$ is downward sloping and in equilibrium equal to the market price. Let the emissions cap be $\overline{E}$.

Lump-Sum Allocation: Grandfathering or Auctioned Permits

With lump-sum allocation, such as grandfathering or auctioned permits, the allocation is invariant to firm behavior. When permits are grandfathered, we assume that this allocation is unconditional and is not affected by decisions to enter or exit the market. Consequently, the choices of emissions rate and output are unaffected by the allocation, at least in a partial equilibrium model.

Let A be the lump-sum allocation to the firm. (With 100% grandfathering, $A = \bar{E}$, while with 100% auctioning, $A = 0$.) Firm profits are

$$\pi^{LS} = (p - c(\mu) - t\mu)y + tA \qquad (1)$$

By the first-order conditions, the firm equalizes the marginal cost of emissions rate reduction with the price of emissions,

$$-c'(\mu) = t \qquad (2)$$

while the market price reflects the unit cost of production and the external cost of the embodied emissions:

$$p = c(\mu) + t\mu \qquad (3)$$

This result corresponds to standard Pigouvian pricing. If each sector's emissions cap were initially set to generate socially optimal reductions, in a market with multiple sectors, trade across sectors would reproduce the same efficient emissions and product pricing—excluding other market failures and general equilibrium effects. With grandfathering, the firm (and its shareholders) receive windfall profits of tA, while auctioning permits raises revenues $t(\bar{E} - A)$.

Updating with Output-Based Allocation

With output-based allocation, the industry receives an allocation A, which is then distributed among the firms in proportion to their output over the relevant period. In equilibrium, the per-unit allocation is $a = A/y$, which a representative competitive firm takes as given. Although we present this representation as a result of an allocation rule with updating in the short run, in practice, the updating time frame may be long or lagged. The key point is that additional output today changes the discount value of the future allocation stream. By the same token, this implicit subsidy can also represent the long run effects of grandfathered permits that are conditional on continued operation.[vi]

The profit function for a representative firm within an industry is expressed as

$$\pi^{OBA} = (p - c(\mu) - t(\mu - a))y \tag{4}$$

The firm's profits are therefore reduced by the cost of the additional permits it must purchase (if $\mu > a$) or increased by the value of excess permits it can sell off (if $\mu < a$). To optimize profits, the firm equalizes the marginal cost of emissions rate reduction as in (2). However, given any emissions rate, the market price is lower than with lump-sum allocation by the value of the per-unit allocation:

$$p = c(\mu) + t(\mu - a) \tag{5}$$

The lower price means that in equilibrium with market demand, output will be higher. Consequently, with output-based allocation, the Pigouvian emissions price and corresponding emissions rate will lead to greater than optimal emissions. The dual to this problem is that to achieve the same level of emissions as the optimal case, the marginal price of emissions must rise and the emissions rate must fall. Thus, for a given amount of emissions reduction, output-based rebating raises the marginal cost of emissions reduction relative to efficient policy.

We next explore this result more formally. Consider two types of permit markets in which this industry might operate. In a restricted market, the industry has its own emissions target. In a broad-based market, the industry participates in a larger cap-and-trade system with other industries, each of which has its own allocation mechanism. An important point is that the subsidy is a function of not the industry average emissions rate but rather the average allocation. In a restricted market, the average emissions rate will equilibrate to equal the average allocation; in a broad-based market, it need not.

Restricted Permit Markets. In restricted permit markets, all the firms participating in a given permit market compete in a single allocation pool. These firms remain price takers both in product and in permit markets. Let us continue to consider representative firms, one for each sector, indexed by *i.* Total emissions for each restricted market are fixed at the level of emissions achieved with Pigouvian pricing, an equilibrium denoted by * (i.e., for sector *i*, $\overline{E}_i = \mu_i^* y_i^*$ where μ_i^* and y_i^* are the optimal emissions rate and output for sector *i*, satisfying Equations (2) and (3) with $t = MB$). Permits totaling $\overline{E}_i$ are then allocated among program participants in each sector according to output shares. The rebate to individual firms in sector *i* thus equals $a_i = \overline{E}_i / y_i$ per unit of output.

Let us denote this equilibrium with superscript *R*. In this case, the average permit allocation equals average emissions in each self-contained permit program, and $a^R{}_i = \mu_i^R$. Thus, $p(y_i^R) = c(\mu_i^R) < c(\mu_i^*) + t\mu_i^* = p(y_i^*)$. Given that $y_i^R > y_i^*$, because of the presence of the output subsidy, to achieve the required emissions level, average emissions rates will have to be lower: $\mu_i^R = \overline{E}_i / y_i^R > \overline{E}_i / y_i^* = \mu_i^*$. As a result, permit prices will be higher, reflecting the higher marginal cost of control: $t_i^R = -c'(\mu_i^R) > -c'(\mu_i^*) = MB$.

Figure 1 depicts the excess burden of output-allocated permits compared with the social optimum in this partial equilibrium setting. The dead-weight loss occurs in two parts: (1) higher-than-optimal production costs, and (2) the damages implied by emissions from the excess production, less the corresponding consumer surplus. In other words, even though total emissions are at their optimal level, the marginal damages from output still exceed the marginal benefits.

It is worth noting that applying separate cap-and-trade programs with output-based allocations to multiple sectors is equivalent to setting performance standards. When the sectoral allocation is based on the emissions corresponding to Pigouvian pricing, as in the case presented here, then marginal abatement costs will diverge according to the elasticity of demand for each sector's output. Alternatively, one could set the performance standards such that marginal abatement costs would be equalized; to replicate this method with OBA requires allocating more permits (compared with the Pigouvian levels) to sectors with relatively more elastic demand. Dissou (forthcoming) simulates this method, using a CGE model to assess the effects of performance standards, set for each sector to both equalize marginal abatement costs and achieve an overall emissions target. A similar method was used by Goulder et al. (1997). Effectively, this represents OBA with a different rule for allocating permits at the sectoral level.

Consequently, modeling performance standards to equalize marginal abatement costs does not represent OBA more generally, since the rule for determining the overall sector's allocation can vary, and it is important. Furthermore, with multiple sectors, it is a far more complicated policy problem to set performance standards so as to equalize marginal abatement costs, whereas markets achieve that with a multisector trading program with OBA.

Multisector Permit Markets. Of course, many pollutants—including greenhouse gases—are emitted from a variety of activities and rarely just a single sector. In this case, allowing permit trading across sectors in a broad-based market can allow for a more efficient allocation of effort. However, output-based allocation of permits can also affect the distribution of effort.

By the same logic as the restricted market model, equilibrium permit prices in the broad market must be higher than optimal, since the output subsidies limit conservation incentives, requiring more emissions rate reduction and higher marginal costs of emissions control. If the sectors are not identical—that is, if they display different cost structures, emissions, or demand elasticities—the implicit subsidies and their effects will vary.

In other words, suppose each sector's targets under restricted permit markets were set so that marginal abatement costs equal marginal benefits with lump-sum allocation, as just previously defined. With OBA, those costs now diverge, as the sector with relatively elastic demand compensate for less conservation by driving down its emissions rate (and driving up its marginal abatement costs) to a greater extent. With trade, marginal abatement costs are again equalized across sectors; the sectors with relatively inelastic demand will tend to increase their overcompliance and sell permits to those whose consumers would otherwise more easily conserve or find substitutes. Furthermore, as costs and thereby output change with multisector trade, the average allocation will not necessarily reflect average emissions in each sector.

To understand the intuition behind these results, consider this simple but extreme example: two sectors compete in a single permit market, each with output-based allocation of permits within the sector, but one sector has perfectly inelastic demand. Let Sector 1 be that sector; its total allocation equals $A_1 = \mu_1^* y_1^*$, and since the equilibrium output level does not change, its average allocation always equals $a_1 = \mu_1^*$. In an equilibrium with multisector emissions trading (denoted with superscript *M*), Sector 1 then has an output price of $p_1^M = c(\mu_1^M) + t^M(\mu_1^M - \mu_1^*)$. Meanwhile, Sector 2 faces more elastic demand. It receives a total allocation of $A_2 = \mu_2^* y_2^*$, which will correspond to an average allocation of $a_2^M = \mu_2^* y_2^* / y_2^M$. The price in that sector then equals $p_2^M = c(\mu_2^M) + t^M(\mu_2^M - \mu_2^* y_2^* / y_2^M)$. In a permit market equilibrium, we know that total emissions across sectors must equal the total cap: $\mu_1^M y_1^* + \mu_2^M y_2^M = A_1 + A_2$. Furthermore, the marginal costs of reducing emissions rates per unit of output must be equalized at the permit price: $-c'(\mu_1^M) = -c'(\mu_2^M)$.

Suppose the emissions price were equal to the optimal marginal abatement cost; whereas Sector 1 always supplies the optimal quantity, in Sector 2, with the output allocation subsidy, a greater quantity will be demanded. The emissions embodied in the extra output would violate the cap, so the permit price must rise and emissions rates fall in both industries to maintain the cap. Then, overall, Sector 1 will emit less than the socially optimal amount, and Sector 2 will emit more.

Alternatively, we can compare this equilibrium to that of restricted permit markets with OBA. With separate permit markets, consumers in the sector with inelastic demand reap the full benefit of the output subsidy, but efficiency is not affected. In Sector 2, efficiency losses are present, since the emissions rate and consumer price are lower than optimal. When these sectors are allowed to trade permits, the permit price would then equilibrate in between, with Sector 1 lowering emissions rates (being more than compensated by the subsidy transfer) and Sector 2 raising them (but still not as high as optimal emissions intensities), so that

$\mu_1^M < \mu_1^*$, $\mu_2^R < \mu_2^M < \mu_2^*$. For Sector 2, lower permit prices and lower control costs mean that consumer prices are even lower than in the restricted permit market case, even though the value of the allocation subsidy falls as well.[vii] Consumer prices in Sector 1 must also be lower; according to the first-order condition for profit maximization, if a firm wants to decrease its emissions rate and trade, its overall costs must be lower.

Taken from another view, output-based allocations can create false gains from trade, based on the extent to which abatement choices are distorted by the output subsidy. Sectors with relatively inelastic demand functions realize a comparative advantage in abatement arising, in a sense, from their greater ability to pass costs along to consumers.

The net effects of permit trading on welfare depend on whether the efficiency loss decreases as it is redistributed. Overcompliance in Sector 1 represents a real resource cost, but it allows Sector 2 to reduce its overcompliance. However, its output price then reflects even less of the cost of the embodied emissions. As the costs of reducing emissions rates are presumably convex, cost savings will arise from spreading overcompliance across the sectors. Thus, the question in the partial equilibrium problem is whether those savings outweigh the additional efficiency loss from more overproduction in Sector 2.

Summary and General Equilibrium Issues

The key tradeoffs between auctioning, grandfathering, and OBA can be divided into efficiency effects and distributional effects, which we summarize in Table. For each policy, the emissions cap determines the environmental benefits, while the allocation method determines the beneficiaries—and to some extent the costs. Auctioning permits benefits the government and taxpayers, since the revenues can expand public goods provision and/or offset other taxes. Grandfathering permits benefits incumbent firms and their shareholders; since marginal cost increases are passed along to consumers in any case, the lump-sum transfer represents windfall profits. However, new entrants are often excluded from these windfalls. Output-based allocation ensures new entrants equal opportunities for allocations, but the mechanism primarily benefits consumers, as the marginal subsidy to firms is passed along in the form of lower prices. Just as the rules for setting individual allocations under grandfathering determine the distribution of windfall profits, the rules for setting the sector-level cap under OBA determine which consumers benefit by setting the effective subsidy for each sector.

From an efficiency perspective, auctioning and grandfathering have the same impact on prices and costs—at least in the absence of other market failures. OBA, by weakening conservation incentives, raises the costs of achieving an emissions target, and in a multi-sector permit market, the overall efficiency loss depends on the distribution of the sector allotments and the effective subsidies.

In a general equilibrium framework, however, these output price effects are more important because of interactions with tax distortions and uncovered sectors. When labor markets are distorted by wage taxes, increases in product prices due to the emissions regulation lower the real wage further, imposing an excess burden. This burden is highest with grandfathering, while auctioning with revenues recycled toward lowering the labor tax minimizes the burden. Since OBA mitigates some of the product price increase, it also mitigates some of the excess burden from the labor market distortion. Similarly, when some sectors remain unregulated, price increases in the regulated sectors encourage consumers to substitute toward unregulated (and/or imported) products, increasing emissions from those sectors. OBA is the only allocation mechanism to mitigate this leakage problem directly, since it affects the relative prices of regulated and unregulated products; revenue recycling, on the other hand, affects only the relative wage rate.

To explore these trade-offs, we next apply this sectoral model of emissions regulation with output-based allocation to a general equilibrium model of the U.S. economy and the case of reducing CO_2.

III. GENERAL EQUILIBRIUM MODEL WITH TRADE AND TAXATION

Description

Since we are primarily concerned with the distributional and efficiency effects of emissions permit allocation mechanisms with taxes and trade, we employ a CGE model from the Global Trade Analysis Project (GTAP), which offers richness in calculating trade impacts. In particular, we can look in detail at the effects on a more diverse and disaggregated set of energy-using sectors than in most climate models.[viii] However, this static model is not designed specifically to study climate policy. It lacks the capability to examine certain issues of import, particularly dynamic responses, since it does not project energy use into the future or allow for technological change. It does, however, allow for capital reallocation. As such, our results should be considered illustrative of short- to medium-term effects (say, 3-5 years, a relatively short perspective for climate policy) on different sectors of implementing a carbon cap-and-trade program using different allocation mechanisms for emissions permits.[ix] Our impacts of interest include CO_2 emissions, production, trade, and employment by sector, as well as overall welfare, both in the United States and abroad, and carbon leakage.

The model and simulations in this paper are based on version 6.1 of the GTAPinGAMS package developed by Thomas Rutherford and documented for version 4 of the dataset and model in Rutherford and Paltsev (2000). The GTAP-EG model serves as the platform for the model outlined here. The GTAP-EG dataset is a GAMS dataset merging version 6 of the GTAP economic data with information on energy flows. A more complete discussion of the energy data used can be found in Complainville and van der Mensbrugghe (1998).

The model is a multi-sector, multi-region general equilibrium model of the world economy as of 2001. Energy requirements and carbon emissions are incorporated into this framework. The production function incorporates most intermediate inputs in fixed proportion, although energy inputs are built into a separate energy nest. Energy production is a CES function nested to three levels. At the lowest level, oil and gas relatively substitutable for one another (elasticity = 2) within the "liquid" nest, while "liquid" energy is less substitutable against coal in the "non-electric nest". Lastly, "non-electric" has low substitutability (0.1) against electricity in the "energy" nest. "Energy" itself has low substitutability (0.5) for the labor-capital composite from the "value-added" nest. Within the "value-added" nest, labor, private capital and public capital have unitary elasticity. Foreign and domestic varieties are substitutable for one another through a standard Armington structure, with the elasticity of substitution between the domestic variety and foreign composite set to half the elasticity of substitution among foreign varieties. The latter elasticities are largely derived from econometrically-based estimates as in Hertel et al. (2004).

Consumption is a composite of goods, services and leisure (further discussion of the labor-leisure choice is below). The energy goods oil, gas and coal enter into final demand in fixed proportions in the "energy" nest, and are unitary elastic with electricity. This composite is then substitutable at 0.5 with other final demand goods and services. Goods and services (including energy) are then substitutable against leisure; the derivation is given below.

Government demand is represented by a similar demand structure and private consumption, with the exception of the labor-leisure component. Government demand is held fixed through all of the experiments, although the funding mechanism (adjustment of a lump-sum tax or the tax on labor) varies as noted below.

Three features are added to the GTAP-EG structure allow us to model the impact of the policy scenarios. First, we add a carbon price. Second, the appropriate structure for simulating an output-based allocation scheme must be incorporated into the model. Third, the household is given a labor-leisure choice so that labor taxes are distorting. This distortionary tax allows us to conduct simulations recycling revenue from pollution permits to offset the distorting tax instrument.

Incorporating Output-Based Allocation. Several changes need to be made to the GTAP-EG code to incorporate output-based allocation of pollution permits. The profit function is not directly accessible in the MPSGE framework. Instead, we incorporate output-based permit allocation through the production function as a sector-based subsidy, combining it with side constraints on the values of a to duplicate the effect on the profit function above. Additionally, we create an additional composite fossil fuel nest to production. This allows us to incorporate the pollution permit as a Leontief technology, allowing us to track pollution permits through the model.

In the original GTAP-EG model, the treatment of energy goods does not allow for tracking of permits by sector. To track pollution permits, we need to ensure that one permit is demanded for each unit of carbon that enters into production. This is accomplished by separating the energy goods into a separate activity, a Leontief technology combining the polluting inputs with permits, into a new composite good (labeled *ffi* in the code, for fossil fuel input). The composite of permit and energy input is then included in the production block for the output good (y), ensuring that the implicit cost of the embodied emissions is reflected in the output price.

The next step is to incorporate the endogenous subsidy implied by the output-based allocation of permits within a sector. We mimic this in the form of an endogenous tax rate, z, into the sector's production function: $z_{i,r} = -t_r \dfrac{A_{i,r}}{y_{i,r}} / p_{i,r}$, where $A_{i,r}$ is the sector-level allocation for sector i of country r.

We consider two potential rules for determining this sector-level allocation. The historical emissions rule defines the sector's apportionment of pollution permits as the baseline unit demand for carbon multiplied by the percentage cap (κ_r) on emissions: $A_{i,r}^{Hist} = \kappa_r \sum_{fe} \chi_{fe,i,r} \phi_{fe,i,r}$. The variable $\chi_{fe,i,r}$ is the carbon coefficient for final energy good $fe \in \{coal, oil, gas\}$ in sector i of country r. The variable $\phi_{fe,i,r}$ represents the demand of the fossil fuel input. The value-added rule apportions the same number of permits based on each of the energy-using sector's share of value added in the base year: $A_{i,r}^{VA} = \sum_j A_{i,r}^{Hist} \frac{VA_{i,r}}{\sum_j VA_{j,r}}$.

The allocation mechanism is active only within those industries or sectors that demand carbon-containing fuel as an intermediate input. Within the GTAP-EG model, this excludes the following sectors: coal; petroleum and coal products; crude oil; natural gas; mining; and dwellings. Final demand for energy products is also subject to emissions permitting. The permits for these activities are freely traded in the same marketplace as those initially allocated based on output. We have a system where all pollution is subject to permitting and all permits are tradable within the country. The difference lies in how permits are distributed in the baseline and how revenues are recycled.

Incorporating Labor-Leisure Choice. The GTAP-EG model has also been extended by incorporating a labor-leisure choice into the household's decision. The procedure is documented in Fox (2002). Incorporating a labor-leisure choice allows us to treat the labor tax as a distorting tax, hence giving us a distorting policy instrument to offset with auction permit revenues. Since we have no data on labor taxes within the GTAP-EG database, we assume a labor tax rate of 40% within Annex B countries and a 20% tax rate within all other countries.[x]

In order to incorporate a labor-leisure choice in the model, it is necessary for us to construct a current level of leisure that is consistent with the output of the U.S. economy and the known characteristics of the U.S. supply of labor. The method presented here relies on the work of Ballard (1999). The consumer utility function is extended by adding a top CES nest allowing the household to substitute consumption of leisure for consumption of goods and services. The top nest of Figure 1 illustrates this structure, with Leisure and the consumption composite combined in a CES function with an elasticity of sig_lsr.

From this top CES nest of the consumer utility function, we derive demand for leisure, and the corresponding expressions for the uncompensated and compensated leisure demand and labor supply elasticities (ε_L and $\varepsilon_L|_{\bar{u}}$).[xi] Ballard (1999) suggests that reasonable values for the supply elasticities are $\varepsilon_L = 0.1$ and $\varepsilon_L|_{\bar{u}} = 0.3$.

Ballard emphasizes that the choice of the time endowment parameter, or how many hours are in a day, determines the total-income elasticity of labor supply. This can have a material impact on the relative responsiveness of changes in tax policy. For example, if the total time endowment is too large relative to the benchmark level of hours worked, the responsiveness of labor supply to a policy change can be implausibly large, despite the fact that the other parameters describing the labor market are well within the range of generally accepted values. To establish the initial value for leisure, we define an additional variable representing the number of hours worked, such that the sum of earned income at the initial wage rate and other (non-wage) income is equal to the benchmark value of final demand. This particular parameterization suggests that leisure is worth about one-quarter of the final demand for goods and services.

Lastly, we establish the benchmark value of the elasticity of substitution between labor and leisure, given econometric estimates of ε_L and $\varepsilon_L|_{\bar{u}}$ and the benchmark level of expenditure on goods and services, as well as the total amount of leisure consumed at the benchmark. The elasticity of substitution varies by country. In the United States, it is 1.736, and it varies between 1.358 and 2.528 elsewhere.

Policy Experiments

The McCain-Lieberman Climate Stewardship Act of 2003 (CSA) proposes to cap emissions in 2010–2016 to 2000 levels, eliminating a decade of increase.[xii] In this spirit, we set the basic policy goal to be a similar 14% reduction of CO_2 emissions from the base-year level (2001 in our case). The CSA also provides for sector-level apportionment to covered emissions sources,[xiii] with broad consideration given to historical emissions, as well as shares to the Climate Change Credit Corporation (CCCC). Details—including the actual shares and the distribution methods within sectors to the firms—are left to future rulemakings by the Commerce Department secretary and the administrator of the Environmental Protection Agency. It is also unclear whether CCCC would use the revenues from permit sales to offer lump-sum rebates (dividends) to consumers, lower federal taxes, or otherwise target the funds. In other words, this overall framework, should it be enacted, seems to offer a wide range of possibilities for allocation; hence, it is important to understand the consequences of alternative mechanisms.

To concentrate on the effects of U.S. program design, we refrain for now from modeling policy changes in other countries, including the European Union. Incorporating the carbon policies under development in other regions would have other general equilibrium effects; however, they are unlikely to change the relative impacts of the U.S. policy scenarios, which we assume are undertaken unilaterally.[xiv]

We conduct four experiments to assess the relative impact of using an output-based allocation scheme compared with other permit allocation methods:

Grandfather: Permits are distributed unconditionally among firms in all sectors (except final demand). This is the equivalent of a lump-sum rebate of all permit revenues.

Auction: All permits are sold—no gratis distribution.

Historical OBA: Allocations to firms are updated based on output shares within their sector. At the sector level, caps are based on historical emissions. Allocations are made in sectors with intermediate energy demand.

Value-Added OBA: Allocations to firms are updated based on output shares within their sector. Sector shares are based on historical shares of value added. Allocations are made in sectors with intermediate energy demand.

In essence, for the gratis distribution scenarios, permit allocation occurs in two phases. First, the rule for allocating the sector's share of the emissions cap is chosen. We consider historical emissions shares and value-added shares as examples. (The sector allocations for these scenarios will be reported in Table 4.) The second phase of permit allocation requires choosing a rule for distributing the sector-level cap among the firms. Our scenarios encompass two options: within each sector, permits are either grandfathered in lump-sum fashion among firms or updated based on output.

In all cases, permits are traded across sectors. Those permits not distributed gratis (i.e., those for final demand in the non-auction scenarios) are auctioned and flow back into the government budget. Furthermore, government revenue is held constant through a labor tax, so any excess revenues from permit sales are recycled to lower the labor tax rate.[xv] By capping the entire economy and allocating to all sectors, we simulate a somewhat more comprehensive carbon trading program than the CSA, which excludes consumers and agriculture.

Results

Perhaps the most striking result is the effect the allocation mechanism has on the permit price, the indicator of marginal abatement cost. Permit prices for the different scenarios are given in

Table 3.

As indicated in the theory, grandfathering and auctioning permits, as lump-sum allocation mechanisms, have nearly identical impacts on the permit price, which we find is about $43/ton C for the 14% reduction. The slight variation occurs due to the general-equilibrium effects of revenue recycling. While the theory predicts that OBA should raise permit prices, this does not hold with any significance for the value-added OBA scenario in general equilibrium, since the implicit subsidies are spread across the economy much like a broad-based tax reduction. However, historical OBA generates a nearly 50% permit price premium compared to the other scenarios. In this case, the implicit subsidy from updating favors large emitters and discourages conservation, thereby requiring the economy to seek reductions elsewhere.

To explore the reasons behind and consequences of these price changes, we divide the other numerical results into two categories: distributional impacts and indicators of efficiency and effectiveness.

Distributional Impacts.

Permit Allocation. Table 4 reports the sector allocations for the gratis allocation options of historical emissions vs. historical value added. In both cases, permits are allocated to sectors with primary energy demand (which tends to exclude primary energy producers from large allocations).xvi Permits representing final demand (residential energy use, representing a little less than 7%) are assumed to be held by the government and auctioned.

Unsurprisingly, the distribution of emissions is quite different from that of the overall economy. The electricity sector accounts for just over two-fifths of national emissions, followed by transport with another quarter, and the chemical industry with nearly a tenth. On the other hand, services represent two-thirds of value added; all other sectors have modest shares—less than 10%.

The distributional effects are interpreted differently if allocations are grandfathered than if they are updated based on output. The GTAP model does not have positive operating profits in equilibrium, assuming instead that average and marginal costs are equalized. As a consequence, the distribution rule does not have allocative effects under grandfathering. Rather, permit allocation shares indicate the distribution of windfall profits by sector to their shareholders (in this case, the representative agent). On the other hand, with OBA, the distribution rule does have allocative effects, but no profit impacts. Although the model does not allow for producer surplus calculations, other variables can serve as indicators of the distributional effects, including sector output and employment. The relative price results will also reflect consumer impacts by sector.

Carbon Emissions. The distribution of effort, in one sense, is represented in Table 5 by the percentage change in emissions across sectors. Nearly all the scenarios had the same impact on the distribution of carbon emissions reductions—with the dramatic exception of historical OBA. In this scenario, emissions reductions shift away from heavy historical polluters—like refining, mining, transport, and other manufacturing—toward other sectors, including agriculture, construction, and particularly final demand.

Surprisingly, we see that several non-energy-intensive sectors have smaller emissions reductions in the Historical OBA scenario. The table also reveals that the sector-specific effects can be much larger than the aggregated effects across broad categories.

Output. Although emissions impacts seem similar across auction, grandfathering, and value-added OBA, the impacts on output do not. For example, non energy intensive sectors benefit from revenue recycling in an auction and from implicit subsidies in VA OBA; however, they contract along with the energy intensive sectors in a grandfathering system. In the case of historical OBA, we see that, corresponding to the emissions impacts, production in historically polluting sectors is significantly higher: the size of the contraction in output is roughly half that in other scenarios. Most notably, electricity production, which falls by 6% in the other scenarios, registers a negligible change with historical OBA, and the transport sector goes from a roughly 8% contraction to a 2% drop.

With this shift from output substitution as a means for emissions reductions in the major polluting sectors, the carbon price rises dramatically to induce reductions elsewhere. This price rise has the added effect of raising the value of the permit allocations, reinforcing the subsidies. For the non energy intensive sectors, the implicit subsidy is outweighed by the higher permit and production costs, lowering output. Several of these sectors also experience greater trade exposure, and in some cases output falls more than with grandfathering.

Employment. The changes in emissions and output have two kinds of impacts on employment by sector. On the one hand, the decrease in emissions leads to a substitution away from energy inputs and toward labor and capital, tending to increase labor demand. On the other hand, a decrease in output tends to decrease labor demand. Furthermore, we will see that the allocation regime affects the after-tax real wage, which also has employment impacts across the economy.

Thus, in employment, we see some sectors increasing their labor demand, while others decrease it, as indicated by the grey cells in Table 7. With auctioned or value-added OBA permits, in general, energy-intensive sectors decrease their demand while non-energy-intensive sectors increase labor demand slightly. With grandfathering, more sectors decrease their labor demand, due to the tax interaction costs which lower the real wage; this is the only scenario in which overall employment falls. With historical OBA, some energy-intensive sectors actually expand employment, while non-energy-intensive sectors decrease theirs, due to the changes in output. In all scenarios, employment in primary energy industries falls significantly, and the allocation regime has relatively little impact.

Efficiency and Effectiveness.

Price Impacts. For the sectors that use energy and electricity as inputs, their own price effects tend to be mirror opposites of the output effects. One puzzle is that under historical OBA, several of the less energy intensive sectors have both lower output and smaller emissions reductions, despite higher permit prices, relative to the other scenarios. The answer lies in the effects of allocation on the different energy prices.

Table 8 reports the changes in prices of energy products. Primary energy prices received by producers are exclusive of the permits required by the downstream users; however, they are affected by the costs of producing the energy good. For example, the petroleum products and (to a lesser extent) natural gas sectors burn fossil fuels in order to make the product, and to this extent, permit requirements are reflected in the producer price. Downstream consumers (or intermediate good producers) face energy costs that include permit costs as well as the producer price.[xvii] Crude oil is only used by the petroleum industry, and consumers of electricity do not have to buy additional permits; those costs become embedded in the electricity price, as do any subsidies from an OBA.

For all primary energy producers, the prices received are lowest and consumer prices highest with historical OBA, due to the higher permit price. The allocation associated with historical OBA also has important impacts on refined fossil fuels and electricity, which are major emitting sectors.

In all but the historical OBA scenario, the price of electricity rises significantly—a signal for other sectors to conserve. With historical OBA, the price actually falls, meaning that electricity is cheaper than without the carbon policy. Correspondingly, more price pressure is placed on other primary energy sources, since more reductions must then come from those sources.

Similarly, with refined petroleum products, in all but historical OBA, producer prices rise, due to the higher production costs associated with abatement and permit requirements.

Those prices then fall with historical OBA, due to the substantial value of the allocation. Indeed, while the producer prices for coal and natural gas fall due to decreased demand, oil consumption is higher in this scenario than in the others. The consumer prices inclusive of permit costs rise less for oil products than for other energy sources (in the other scenarios, the oil price rise is roughly 90% of the natural gas price rise; with historical OBA, it is 70% of the natural gas price rise). This relative price change makes oil more attractive—despite its higher emissions content—resulting in a shift toward oil (or less of a reduction in oil consumption). This shift explains the lesser pressure on the price of crude oil.

More importantly, this relative price change dominates the effect of higher permit prices in the incentives for the non energy intensive sectors, as their shift from gas to oil increases their emissions intensity under historical OBA relative to the other scenarios (see Table 5). As a result, with almost all of the producing sectors reducing their emissions abatement, either due to the subsidy or the relative price change, final demand ends up shouldering a disproportionate share of the burden in the historical OBA scenario.

Trade. Since historical OBA causes the greatest distortion in relative prices, it has the greatest impact on trade. Table 9 presents the change in net exports, evaluated at the base year prices, in millions of dollars. The net export position of the heavy emitters falls much less dramatically, and it even rises for electricity and for energy intensive sectors overall. The chemicals industry benefits particularly from lower petroleum prices. However, some sectors that are relatively more competitive in other allocation regimes see their net exports fall with historical OBA, namely other industries and services. These sectors face higher permit prices and labor costs and little subsidies.

Net exports of primary energy products increase in all scenarios, since domestic demand declines; the exception is that increased electricity production with historical OBA increases coal imports. Overall, net exports fall most with historical OBA and least with grandfathering, in part since domestic consumption is lowest in the latter scenario.

For the most part, these changes amount to less than 1% of production, with the exception of some energy industries and mining. The decline in net exports of natural gas and mining represented 2-3% of production in those sectors in all but the historical OBA scenario. The increase in crude oil net exports represents 17% of production in the historical OBA scenario and 23% of production in the others, due to the drop in domestic consumption. Overall, however, little difference emerges among the scenarios, as total net exports fall by 0.03% of the total value of production; on the other hand, the carbon leakage profiles do vary.

Carbon Leakage. Carbon leakage is driven primarily by the relative price effects for energy intensive sectors. Since historical OBA is the only scenario to target those sectors specifically, it proves the most effective at limiting the increase in emissions among trading partners, with 12.5% of domestic reductions offset be increases abroad, compared to 15.4% with the other scenarios. Value-added OBA, grandfathering, and auctioned permits have nearly identical impacts on leakage. Thus, while OBA has the potential to reduce carbon leakage relative to other methods, the rule for allocating at the sector level is important for determining this effect.

Table 10 compares leakage by sector, focusing on the two scenarios of auctioning and historical OBA, since the others have such similar results to allocation by auction. A major impact of historical OBA is felt in the reduction of the leakage rate for refined products, chemicals, and transportation. However, this effect is partly offset by fewer reductions at home. Leakage pressure is increased for other industries and services, some of which have very high rates, but they compose a smaller share of total emissions reductions.

On the other hand, primary energy sectors exhibit strongly negative leakage. Since the U.S. reduces consumption of primary energy, imports fall and so do foreign emissions. Emissions related to producing primary energy products are on a smaller order of magnitude than those from burning those products, but the foreign reductions are as (or more) significant as the domestic ones in that sector. As such, the data seem to indicate that foreign primary energy production (particularly of coal) is more emissions intensive than production at home. The shift in U.S. demand also seems to increase emissions from foreign final demand, due to downward pressure on global energy prices.

Table 11 shows carbon leakage by trading partner for each of the allocation scenarios. An important caveat is that we do not model an emissions cap in other countries, like the EU, which would tend to limit leakage to those trading partners. For example, if the rest of the Annex I parties were to adhere to their Kyoto emissions caps, then more than half of the leakage concern of a U.S. cap would be eliminated.

Summary Economic Indicators for the United States. The relative efficiency of the allocation policies is reflected in the change in the summary economic indicators, reported in Table 12. The primary indicator is welfare, which is measured in equivalent variation. Change in total production is another indicator of economic impacts.[xviii] Impacts on workers, consumers, and tax payers are reflected by changes in employment, the real wage, and the labor tax rate.

Trade impacts are indicated by the overall change in net exports and in emissions leakage. The emissions permit price reflects the marginal cost of abatement.

Overall, the 14% reduction induces less than a 1% change in the summary economic indicators for all the scenarios. These impacts are consistent with the range found by other climate models.[xix] Although the changes may seem small, the relative effects of the allocation scenarios are still illustrative.

As predicted, auctioning permits with revenue recycling produces the smallest welfare loss for the United States, measured in equivalent variation. In fact, the welfare impact is slightly positive, implying a small double-dividend effect, as this mechanism leads to an increase in the real wage and employment, due to the significant fall in the labor tax rate. In other words, the implicit energy tax is less distorting on the margin than the existing labor tax.

Grandfathering permits entails the largest welfare cost—and the largest drop in the real wage—since the loss of tax revenues from the economic contraction requires an increase in the labor tax rate.[xx]

The most notable effects of historical OBA are the dramatic rise in the price of permits, which are a 50% more costly than all of the other scenarios, and the fall in the leakage rate, which is three-quarters that of the others. The revenue adjustments were minor—even slightly negative, because of the greater value of the permits withheld for auction.[xxi] The impacts on overall welfare are closer to those with grandfathering than with auctions; however, historical OBA had the smallest decrease in production, and little impact on the real wage. In other words, the mitigation of the consumer price increases, easing the burden of tax and trade distortions, roughly offset the inefficiencies in allocating emissions reductions.

Value-added OBA functions a good deal like a consumption tax reduction and therefore approaches auctioning in efficiency (though not perfectly so, since not quite all sectors generating value added receive allocations). Overall, the welfare cost was only slightly larger than auctioning. This scenario saw a slight increase in the real wage, due not only to a small drop in the tax rate but also to a more even distribution of the effective output subsidies throughout the economy.

It is also interesting to note the effects of U.S. emissions permit allocation on global welfare, although the magnitudes are admittedly small in percentage terms (less than 0.3% of GDP for any given region). The overall international impacts of U.S. climate policy are presented in Table 13.

We see that the small welfare increase in the U.S. with auctioned permits is outweighed by losses abroad, implying the double dividend does not hold overall. And although historical OBA eases some of the burden of tax and trade distortions compared with grandfathering for the United States, this set of output subsidies seems to have the strongest impact on the welfare of trading partners; it ranks the lowest in terms of global welfare.

Sensitivity Analysis. The relative efficiency of OBA is obviously sensitive to the rule for distributing the permits at the sector level. However, it is also sensitive to the degree of the tax distortion to the labor supply. Obviously, at the extreme, if labor supply were perfectly inelastic, labor and consumption taxes would have no distorting effect, grandfathering permits would be equivalent to auctioning, and any OBA would be less efficient. Therefore, there should be some labor supply elasticity at which the benefit from using output subsidies to mitigate tax interactions is outweighed by the cost in terms of distorting relative prices. To better assess the role of the labor-leisure tradeoff in determining the relative efficiency of the allocation scenarios, we conduct sensitivity analysis with respect to the elasticity of labor supply.

Figure 3 illustrates the welfare changes arising from different combinations of the compensated and uncompensated labor supply elasticities. Both the absolute and relative elasticities are important, since the value placed on leisure rises as the difference between the two elasticities increases (Ballard 1999). The combination (2,3) is closest to our baseline values of (0.2, 0.29) for compensated and uncompensated elasticities, respectively. We see that more inelastic labor supply compresses the differences between the policies, while higher elasticities make them more pronounced. Within this range, however, the rankings remain the same.

Another question regards the relationship between the ranking of the policy options and the stringency of the target.

Figure 4 reveals that relative costs are not proportional to the emissions reduction target. In these simulations, using OBA with the historical emissions rule has some benefits relative to grandfathering for targets up to an 18% reduction; for more stringent emissions caps, however, the distortions created by the corresponding subsidies outweigh the benefits, and historical OBA becomes increasingly the most costly option.

Finally, we also conducted analysis for *net* emissions targets, such that each policy would achieve the same amount of emissions reductions globally, net of leakage. The results are very much the same; despite historical OBA's superiority in mitigating leakage, that policy is still dominated by auctioning and value-added OBA for all reduction targets. The net target does extend the range over which historical OBA is preferred to grandfathering, with those welfare costs crossing at a target of around a 21% reduction.

IV. CONCLUSION

The use of emissions trading represents an important step in improving the efficiency of environmental regulation. However, the tremendous implicit value of the capped emissions—particularly in the case of carbon—raises important political and economic questions about how to allocate the permits. The practical reality seems that the vast majority of permits will be given away gratis to the regulated industries. If so, can we design the allocation process to mitigate the problems of welfare costs, tax distortions, and carbon leakage?

The answer may be that these goals pose trade-offs. In terms of the overall economic indicators—welfare, production, employment, and real wages—auctioning with revenue recycling is the preferred allocation method. Value-added OBA, which effectively attempts to embed the proportional tax rebate into consumer prices, is a fairly close second by these metrics, improving notably over lump-sum grandfathering.

However, in terms of mitigating carbon leakage, historical OBA is clearly the most effective. For the same reason—that it limits price rises in energy-intensive sectors—it also poses the greatest costs on other sectors. While this result would imply important efficiency losses relative to grandfathering in a partial equilibrium model, we find that these losses are offset by gains in terms of mitigating tax interactions in a general equilibrium framework. However, for more stringent targets, historical OBA can indeed become more costly in welfare terms.

This raises the issue of whether the sector allocation rule can be optimized to target some set of these goals. For example, what might the optimal subsidies to limit leakage look like, and what impact might they have on overall welfare?[xxii] What are the relative roles of carbon intensity, trade exposure, and demand elasticities in determining these subsidies?

Although theory offers some support that OBA can enhance the economic efficiency and environmental additionality of emissions trading in a second-best setting, the question remains whether OBA can pass legal muster under world trade rules.[xxiii] From an economic point of view, taxing the carbon content of imports from countries with lesser climate policies can similarly combat leakage; however, such an import tax is very likely to be challenged in the World Trade Organization. Since allocation is perceived as a component of environmental regulation, not a direct subsidy, OBA may enjoy legal leeway. On the other hand, since OBA can create a significant subsidy to industry, it has the potential for abuse in practice. Indeed, unlike with sector-specific performance standards or emissions trading systems, with broad-based, intersectoral trade, OBA can be designed to offer subsidies that outweigh the direct effect of the regulatory compliance costs. Resolving the question of whether OBA is a legal policy tool (and under what conditions) could have important implications for the efficiency—and inefficiency—of future climate policies.[xxiv]

Overall, however, we find relatively small magnitudes for the welfare costs of the policies. To put in some perspective, since climate change may increase the risk of extreme weather events, the annual welfare costs of the policies we simulate range to the U.S. from about 0-3% of the infrastructure and property losses from Hurricane Katrina.[xxv] Although the estimated differences among the policies may seem small in relative terms, that does not mean policy makers need not be judicious in designing an emissions cap and trade program for carbon. For one, we have modeled a broad-based policy with complete coverage of the economy; to the extent that an actual policy will cover only a restricted number of sectors, domestic leakage can be as much a concern as carbon leakage abroad, and the same overall reduction target will be costlier to meet. Furthermore, over time, targets will need to be more stringent, and incentives for technological change will be more important, meaning policy choices that become embedded in national carbon emissions regulation will have greater consequences.

References

Babiker, M.H., G.E. Metcalf, and J. Reilly. 2001. Distortionary Taxation in General Equilibrium Climate Modeling. Paper prepared for the Fourth Annual Conference on Global Economic Analysis, Purdue University, W. Lafayette, IN.

Bernard, A., C. Fischer, and A. Fox. Forthcoming. Is There a Rationale for Rebating Environmental Levies? *Resource and Energy Economics*.

Bovenberg, A.L., and L.H. Goulder. 2001. "Neutralizing the Adverse Industry Impacts of CO2 Abatement Policies: What Does It Cost?" in C. Carraro and G. Metcalf, eds., *Behavioral and Distributional Effects of Environmental Policies*, University of Chicago Press.

Burton, M.L. and M.J. Hicks. 2005. Hurricane Katrina: Preliminary Estimates of Commercial and Public Sector Damages. Report. Center for Business and Economic Research Marshall University, Huntington, WV. (September).

Burtraw, D., K. Palmer, R. Bharvirkar, and A. Paul. 2002. The Effect on Asset Values of the Allocation of Carbon Dioxide Emission Allowances. *The Electricity Journal* 15 (5): 51-62.

Complainville, C., and D. van der Mensbrugghe. 1998. Construction of an Energy Database for GTAP V4: Concordance with IEA Energy Statistics. Manuscript. Paris: OECD Development Centre. April.

Dissou, Y. (forthcoming). Cost-effectiveness of the Performance Standard System to Reduce CO_2 Emissions in Canada: A General Equilibrium Analysis. *Resource and Energy Economics.*

EIA. 2003. Analysis of S.139, the Climate Stewardship Act of 2003. SR/OIAF/2003-02. Energy Information Administration, Office of Integrated Analysis and Forecasting, U.S. Department of Energy, Washington, DC.

Fischer, C. 2003. "Combining Rate-Based and Cap-and-Trade Emissions Policies," *Climate Policy* 3S2: S89-S109.

Fischer, C., and R.D. Morgenstern. 2003. Carbon Abatement Costs: Why the Wide Range of Estimates? RFF Discussion Paper 03-42. Washington, DC: Resources for the Future.

Fischer, C., S. Hoffmann, and Y. Yoshino. 2003. Multilateral Trade Agreements and Market-Based Environmental Policies. In J. Milne, K. Deketelaere, L. Kreiser, H. Ashiabor, eds., *Critical Issues in International Environmental Taxation: International and Comparative Perspectives,* vol. 1. Richmond, UK: Richmond Law and Tax Ltd.

Fox, A.K. 2002. Incorporating Labor-Leisure Choice into a Static General Equilibrium Model. Working paper. Washington, DC: U.S. International Trade Commission.

Fullerton, D., and G. Metcalf. 2001. Environmental Controls, Scarcity Rents, and Pre-Existing Distortions. *Journal of Public Economics* 80: 249–267.

Goulder, L.H. 1995. Environmental Taxation and the "Double Dividend": A Reader's Guide. *International Tax and Public Finance* 2: 157–83.

Goulder, L.H., I.W.H. Parry, and D. Burtraw. 1997. Revenue-Raising vs. Other Approaches to Environmental Protection: The Critical Significance of Pre-Existing Tax Distortions. *RAND Journal of Economics* 28(4): 708–31. Winter.

Goulder, Lawrence H., ed. 2002. *Environmental Policy Making in Economies with Prior Tax Distortions*, Edward Elgar.

Goulder, Lawrence H., Ian Parry, Roberton Williams III, and Dallas Burtraw. 1999. "The Cost-Effectiveness of Alternative Instruments for Environmental Protection in a Second-Best Setting", *Journal of Public Economics* 72(3):329-360, 1999.

Hertel, Thomas, David Hummels, Maros Ivanic, and Roman Keeney. 2004. "How Confident Can We Be in CGE-Based Assessments of Free Trade Agreements?" GTAP Working Paper No. 26, Revised Version, March, 2004.

Jensen, J., and T.N. Rasmussen. 2000. Allocation of CO_2 Emission Permits: A General Equilibrium Analysis of Policy Instruments. *Journal of Environmental Economics and Management*, 40: 111-136.

Paltsev, Sergey, John M. Reilly, Henry D. Jacoby, A. Denny Ellerman and Kok Hou Tay. 2003. Emissions Trading to Reduce Greenhouse Gas Emissions in the United States: The McCain-Lieberman Proposal, MIT Joint Program on the Science and Policy of Global Change, Report 97, Cambridge, MA

Parry, I.W.H. 1995. Pollution Taxes and Revenue Recycling. *Journal of Environmental Economics and Management* 29: S64–77.

Parry, Ian W.H. and Roberton C. Williams. 1999. Second-Best Evaluation of Eight Policy Instruments to Reduce Carbon Emissions, *Resource and Energy Economics*, 21: 347-373.

Rutherford, T.F., and S.V. Paltsev. 2000. GTAPinGAMS and GTAP-EG: Global Datasets for Economic Research and Illustrative Models. Working paper. Boulder, CO: Department of Economics, University of Colorado. September.

Smith, Anne E., Paul Bernstein, and W. David Montgomery. 2003. The Full Costs of S.139, With and Without Its Phase II Requirements, report prepared for the Tech Central Science Foundation (October 27). Washington, DC: Charles River Associates. Available at http://www.crai.com/pubs/pub_3694.pdf.

Smith, Anne E. 2004. Alternatives to Mitigate the Economic Impacts of the McCain-Lieberman Bill. Prepared for The Climate Policy Center (December 9). Washington, D.C.: Charles River Associates.

Appendix

MPSGE Code for Output-Based Allocation

Fossil fuel production activity (crude, gas and coal):

```
$prod:y(xe,r)$vom(xe,r)  s:(esub_es(xe,r))  id:0

o:py(xe,r) q:vom(xe,r) a:gov(r) t:ty(xe,r)

      i:pa(j,r)$(not fe(j)) q:vafm(j,xe,r) p:pai0(j,xe,r) a:gov(r)

+t:ti(j,xe,r) id:

      i:pl(r) q:(ld0(xe,r)/pl0(r)) p:pl0(r) a:gov(r) n:ltax(r) id:

 i:pr(xe,r) q:rd0(xe,r)

i:pffi(fe,xe,r)$vafm(fe,xe,r)q:(vafm(fe,xe,r)*pai0(fe,xe,r)) id:
```

Non-fossil fuel production (includes electricity and refining):

```
$prod:y(i,r)$nr(i,r)   s:0  vae(s):0.5  va(vae):1

+                       e(vae):0.1  nel(e):0.5 lqd(nel):2

+                       oil(lqd):0 col(nel):0 gas(lqd):0

o:py(i,r) q:vom(i,r)  a:gov(r) t:ty(i,r) a:ra(r)

+n:z(i,r)$(cap(i) and subject(r) and sum(fe,vafm(fe,i,r)))

i:pl(r) q:(ld0(i,r)/pl0(r)) p:pl0(r) a:gov(r) n:ltax(r) va:

i:rkr(r)$rsk q:kd0(i,r) va:

i:rkg$gk q:kd0(i,r) va:
```

```
i:pa(j,r)$(not fe(j)) q:vafm(j,i,r) p:pai0(j,i,r) e:$ele(j)

+a:gov(r) t:ti(j,i,r)

i:pffi(fe,i,r)$vafm(fe,i,r) q:(vafm(fe,i,r)*pai0(fe,i,r)) fe.tl:
```

Fossil fuel composite (fuel-plus-permit):

```
$prod:ffi(fe,i,r)$(vafm(fe,i,r)) s:0

o:pffi(fe,i,r) q:(vafm(fe,i,r)*pai0(fe,i,r))

i:pcarb(r)#(fe)$(cap(i) and subject(r)) q:carbcoef(fe,i,r) p:1e-6

i:pa(fe,r) q:vafm(fe,i,r) p:pai0(fe,i,r) a:gov(r) t:ti(fe,i,r)
```

OBA-related side constraints:

```
$constraint:z(i,r)$(cap(i) and subject(r) and sum(fe,vafm(fe,i,r)))

z(i,r)*py(i,r)*y(i,r)*vom(i,r) =e= (-ebar(i,r)*y(i,r))*pcarb(r);


$constraint:ebar(i,r)$(cap(i) and subject(r) and sum(fe,vafm(fe,i,r)))

ebar(i,r) * y(i,r) =e= PctCap(r) * (OBA_Hst * alloc_hst(i,r) +

OBA_Va * alloc_VA(i,r)) ;
```

Table 1: Summary of Allocation Mechanism Effects

Allocation Mechanism	*Distributional Effects*	*Efficiency Effects*
Auctioning (with revenue recycling)	Benefits taxpayers; Permit costs are passed on to consumers	Revenue recycling mitigates fall in real wage, though high prices in regulated sectors can encourage emissions leakage
Grandfathering (lump-sum allocation)	Gives windfall rents to incumbent firms and their shareholders; no allocation for expanded production or new entrants; Permit costs are passed on to consumers	High prices interact with labor tax distortions and encourage emissions leakage
Output-based allocation (updating)	Benefits consumers with smaller price increases; Allocations guaranteed for expanded production or new entrants	Smaller product price increases mean less interaction with labor taxes and less emissions leakage; However, conservation is discouraged and marginal abatement costs are higher; Relative prices among regulated sectors are also distorted

Table 2: Allocation Policy Scenarios

	Auction	*Grandfather*	*Historical OBA*	*VA OBA*
Sector-level allocation	None	May be based on historical emissions, value added, or other	Based on historical emissions	Based on historical value added
Within-sector allocation to firms	None	Lump-sum	Updated based on output shares	Updated based on output shares

Table 3: Emissions Price (2001 dollars per ton C)

Auction	*Grandfather*	*Hist. OBA*	*VA OBA*
$ 49.21	$ 48.59	$ 70.47	$ 49.87

Table 4: Sector Shares of Carbon Cap with Historical and Value-Added Rules

Sector	*Historical*	*Value-Added*
Electricity	*47.6%*	*1.5%*
Petroleum and coal products (refined)	9.7%	0.1%
Chemical industry	7.6%	3.0%
Other mining and metals	1.6%	1.2%
Paper-pulp-print	1.0%	1.9%
Iron and steel industry	0.8%	0.6%
Emissions intensive	*20.7%*	*6.7%*
Agriculture	1.0%	1.2%
Food products	1.0%	2.8%
Transport equipment	0.3%	2.0%
Other machinery	0.3%	4.2%
Other manufacturing	0.3%	3.1%
Textiles-wearing apparel-leather	0.2%	1.0%
Wood and wood-products	0.2%	1.0%
Other industry	*3.3%*	*15.2%*
Natural gas	1.2%	0.1%
Crude oil	0.4%	0.0%
Coal	0.0%	0.1%
Primary energy	*1.6%*	*0.2%*
Other services	2.5%	50.0%
Trade, wholesale and retail	1.4%	16.3%
Construction	0.2%	6.7%
Services (excl. transport)	*4.1%*	*73.0%*
Transport	*22.9%*	*3.4%*
Total permits allocated	**1,306,099**	**1,306,099**

Table 5: Percentage Change in Emissions

Sector	*Auction*	*Grandfather*	*Hist. OBA*	*VA OBA*
Electricity	*-19.9*	*-19.9*	*-19.2*	*-19.9*
Petroleum and coal products (refined)	-13.5	-13.5	-11.9	-13.6
Chemical industry	-9.7	-9.7	-8.7	-9.6
Other mining and metals	-29.2	-29.2	-24.8	-28.9
Paper-pulp-print	-9.0	-9.0	-8.1	-8.9
Iron and steel industry	-7.8	-7.8	-7.0	-7.7
Energy intensive	*-11.6*	*-11.6*	*-10.4*	*-11.6*
Agriculture	-10.5	-10.6	-12.3	-10.4
Food products	-9.3	-9.4	-8.9	-9.2
Transport equipment	-6.9	-6.9	-5.2	-6.7
Other machinery	-6.5	-6.5	-4.8	-6.3
Other manufacturing	-6.3	-6.3	-4.5	-6.1
Textiles-wearing apparel-leather	-7.8	-7.9	-6.4	-7.7
Wood and wood-products	-6.6	-6.5	-4.6	-6.4
Other industry	*-7.6*	*-7.7*	*-7.4*	*-7.6*
Natural gas	-7.2	-7.3	-9.0	-7.4
Crude oil	-8.8	-8.9	-8.0	-9.0
Coal	-24.7	-24.6	-25.2	-24.8
Primary Energy	*-7.7*	*-7.8*	*-8.8*	*-7.9*
Other services	-12.2	-12.4	-7.6	-11.9
Trade, wholesale and retail	-6.2	-6.3	-3.9	-6.0
Construction	-10.1	-10.0	-11.1	-10.0
Services (excl. transport)	*-0.1*	*-0.1*	*0.0*	*-0.1*
Transport	*-32.3*	*-32.3*	*-30.4*	*-32.1*
Final Demand	*-8.9*	*-9.0*	*-13.1*	*-9.1*
Total	**-14.0**	**-14.0**	**-14.0**	**-14.0**

Table 6: Percentage Change in Output

Sector	*Baseline value (billions of $)*	*Auction*	*Grandfather*	*Hist. OBA*	*VA OBA*
Electricity	*258,050*	*-5.8*	*-5.9*	*-0.4*	*-5.8*
Petroleum & coal products (refined)	145,767	-10.9	-10.9	-8.2	-10.9
Chemical industry	716,372	-1.9	-2.0	-0.7	-1.9
Other mining	272,266	-4.0	-4.3	-2.0	-4.1
Paper-pulp-print	391,641	-0.3	-0.4	-0.3	-0.3
Iron and steel industry	142,544	-0.8	-0.9	-0.8	-0.8
Energy intensive	*1,668,589*	*-2.12*	*-2.24*	*-1.26*	*-2.14*
Agriculture	221,217	-0.2	-0.4	-0.4	-0.2
Food products	744,582	0.0	-0.2	-0.1	0.0
Transport equipment	661,162	-0.1	-0.2	-0.3	-0.1
Other machinery	787,603	0.0	-0.1	-0.7	0.0
Other manufacturing	705,388	-0.1	-0.3	-0.7	-0.1
Textiles-wearing apparel-leather	270,617	-0.3	-0.6	-0.4	-0.4
Wood and wood-products	227,138	-0.2	-0.3	-0.3	-0.2
Other industry	*3,617,707*	*-0.09*	*-0.26*	*-0.44*	*-0.11*
Natural gas	49,657	-5.3	-5.3	-6.6	-5.4
Crude oil	39,395	-5.0	-5.0	-4.5	-5.1
Coal	32,630	-16.1	-16.0	-16.5	-16.2
Primary Energy	*121,682*	*-8.08*	*-8.09*	*-8.55*	*-8.19*
Other services	7,051,427	0.3	-0.1	-0.2	0.2
Trade, wholesale and retail	2,456,004	0.1	-0.1	-0.1	0.0
Construction	1,351,225	-0.1	-0.2	-0.1	-0.1
Dwellings	758,281	0.5	0.3	-0.1	0.0
Services (excl. transport)	*11,616,937*	*0.10*	*-0.07*	*-0.11*	*0.05*
Transport	*670,410*	*-7.8*	*-8.2*	*-2.0*	*-8.0*
Total[xxvi]	**17,953,375**	**-0.38**	**-0.54**	**-0.36**	**-0.42**

Table 7: Percentage Change in U.S. Labor Demand

Sector	*Baseline Labor Demand (billions $)*	*Auction*	*Grandfather*	*Hist. OBA*	*VA OBA*
Electricity	*29.5*	*0.1*	*-0.3*	*7.8*	*0.0*
Petroleum & coal products (refined)	2.1	-2.3	-2.6	2.1	-2.5
Chemical industry	128.3	-0.7	-1.0	0.3	-0.8
Other mining	58.7	-0.4	-0.6	-0.1	-0.5
Paper-pulp-print	91.4	0.2	-0.1	0.0	0.1
Iron and steel industry	34.8	0.2	0.0	-0.3	0.1
Energy intensive	*315.2*	*-0.3*	*-0.6*	*0.1*	*-0.4*
Agriculture	34.1	0.2	-0.1	0.1	0.1
Food products	103.9	0.3	0.0	0.2	0.2
Machinery	355.3	0.1	-0.1	-0.4	0.1
Other manufacturing	177.3	0.1	-0.2	-0.6	0.0
Textiles-wearing apparel-leather	58.3	0.0	-0.4	-0.3	-0.1
Wood and wood-products	47.6	0.2	-0.1	-0.1	0.1
Other industry	*776.4*	*0.2*	*-0.1*	*-0.3*	*0.1*
Natural gas	9.2	-7.2	-7.3	-9.0	-7.4
Crude oil	3.3	-8.8	-8.9	-8.0	-9.0
Coal	6.8	-24.7	-24.6	-25.2	-24.8
Primary energy	*19.3*	*-13.7*	*-13.7*	*-14.5*	*-13.8*
Services	2,689.8	0.3	0.0	0.0	0.2
Trade, wholesale and retail	945.2	0.3	0.0	0.0	0.2
Construction	418.9	-0.1	-0.2	0.0	-0.1
Dwellings	5.7	0.7	0.3	0.1	0.1
Services (excl. transport)	*4,059.6*	*0.2*	*0.0*	*0.0*	*0.1*
Transport	*184.6*	*-0.4*	*-0.6*	*1.8*	*-0.5*
Total	**5,384.60**	**0.12**	**-0.15**	**0.00**	**0.03**

Table 8: Emissions Price and Percentage Change in Energy Prices

Sector	*Auction*		*Grandfather*		*Hist. OBA*		*VA OBA*	
Permit Cost	*Excl.*	*Incl.*	*Excl.*	*Incl.*	*Excl.*	*Incl.*	*Excl.*	*Incl.*
Petroleum & coal products (refined)	2.5	18.8	2.5	18.6	-2.2	22.3	2.6	19.1
Natural gas	-3.2	21.6	-3.1	21.4	-3.8	32.4	-3.1	22.1
Coal	-14.9	77.8	-14.8	76.7	-15.6	118.0	-14.8	79.1
Crude oil	-4.0		-3.9		-3.3		-4.0	
Electricity	9.4		9.3		-1.5		9.5	

Table 9: Change in Net Exports
(millions of 2001 dollars)

Sector	*Auction*	*Grandfather*	*Hist. OBA*	*VA OBA*
Electricity	*-629*	*-615*	*47*	*-639*
Petroleum and coal products (refined)	-551	-506	1,739	-593
Chemical industry	3,470	3,467	4,323	3,360
Other mining and metals	-7,958	-7,705	-2,539	-7,917
Paper-pulp-print	-461	-432	-439	-447
Iron and steel industry	-1,464	-1,448	-286	-1,475
Energy intensive	*-6,963*	*-6,624*	*2,799*	*-7,071*
Agriculture	-357	-280	-630	-307
Food products	-309	-216	-676	-262
Transport equipment	-450	-545	-1,773	-492
Other machinery	1,120	861	-3,358	1,141
Other manufacturing	-210	-406	-3,367	-227
Textiles-wearing apparel-leather	-494	-365	-912	-475
Wood and wood-products	-139	-130	-404	-125
Other industry	*-839*	*-1,082*	*-11,119*	*-746*
Natural gas	-861	-850	-282	-861
Crude oil	9,103	9,109	6,546	9,086
Coal	20	19	-24	21
Primary Energy	*8,261*	*8,278*	*6,240*	*8,246*
Other services	1,823	1,755	-1,602	1,862
Trade, wholesale and retail	171	179	-201	174
Construction	-6,435	-6,302	-1,348	-6,498
Services (excl. transport)	*-4,441*	*-4,368*	*-3,152*	*-4,463*
Transport	*-129*	*-127*	*-60*	*-130*
Total	**-4,741**	**-4,538**	**-5,245**	**-4,803**

Table 10: Carbon Leakage in the Largest Emitting Sectors

Sector	Auction (Grandfather, VA OBA similar)			Historical OBA		
	Reductions, 1000 MT C	*leakage, % of sector reductions*	*leakage, % of total leakage*	*Reductions, 1000 MT C*	*leakage, % of sector reductions*	*leakage,% of total leakage*
Electricity	*121,366*	*10*	*35.2*	119,634	*10*	*41.6*
Petroleum & coal products (refined)	18,691	44	22.5	14,936	25	13.8
Chemical industry	9,237	37	10.5	8,490	31	9.5
Other mining and metals	2,509	49	3.8	2,614	46	4.4
Paper-pulp-print	1,126	14	0.5	1,050	18	0.7
Iron and steel industry	755	79	1.9	696	105	2.7
Energy Intensive	*32,317*	*42*	*39.2*	27,786	*31*	*31.0*
Agriculture	1,263	30	1.2	1,555	29	1.6
Food products	1,115	13	0.5	1,103	18	0.7
Transport equipment	288	12	0.1	228	22	0.2
Other machinery	234	30	0.2	180	82	0.5
Other manufacturing	207	65	0.4	153	150	0.8
Textiles-wearing apparel-leather	232	39	0.3	198	63	0.5
Wood and wood-products	171	10	0.1	126	21	0.1
Other industry	*3,510*	*25*	*2.8*	3,542	*34*	*4.4*
Natural gas	1,280	-118	-3.9	1,580	-100	-5.7
Crude oil	497	-285	-3.7	409	-286	-4.3
Coal	23	-965	-0.7	24	-1129	-1.0
Primary energy	*1,800*	*-178*	*-8.3*	*2,013*	*-150*	*-11.0*
Other services	1,965	21	1.3	1,399	42	2.1
Trade, wholesale and retail	1,043	15	0.5	672	31	0.8
Construction	279	38	0.3	319	38	0.4
Services (excl. transport)	*3,287*	*20*	*2.1*	*2,390*	*38*	*3.3*
Transport	*35,289*	*16*	*15.4*	*30,206*	*12*	*12.8*
Final Demand	*21,760*	*20*	*13.7*	*33,758*	*14*	*17.8*
Total	**219,330**	**15.4**	**100.0**	**219,330**	**12.5**	**100.0**

Table 11: Carbon Leakage as Percentage of U.S. Reduction

Country	*Auction, Grandfather, VA OBA*	*Hist. OBA*
Europe	4.3	3.4
Canada	1.2	0.7
Japan	1.0	0.9
Other OECD	0.4	0.3
Former Soviet Union	1.0	0.9
Central European Associates	0.6	0.5
Annex I	*8.4*	*6.7*
China (incl. HK & Taiwan)	1.9	1.7
India	0.6	0.6
Brazil	0.4	0.3
Other Asia	1.4	1.2
Mexico + OPEC	1.0	0.7
Rest of World	1.6	1.3
Total	**15.4**	**12.5**

Central European Associates include Bulgaria, Czech Republic, Hungary, Poland, Romania, Slovakia, and Slovenia.

Table 12: Summary Indicators for the United States

Indicator	*Unit*	*Auction*	*Grandfather*	*Hist. OBA*	*VA OBA*
Welfare	% change in equivalent variation	0.00	-0.05	-0.04	-0.01
Production	% change	-0.38	-0.54	-0.36	-0.42
Employment	% change	0.12	-0.15	0.00	0.03
Real wage	% change	0.46	-0.59	-0.02	0.09
Labor tax change	percentage pts	-1.81	0.00	-0.34	-0.40
Carbon leakage	% of reductions	15.4	15.3	12.5	15.3
Permit price	$/metric ton C	$49.21	$48.59	$70.47	$49.87

Table 13: Change in Global Welfare
(Equivalent Variation; Millions of USD)

Country	*Auction*	*Grandfather*	*Hist. OBA*	*VA OBA*
United States	443	-4,582	-3,814	-1,202
Rest of World	-579	-380	-1,595	-661
World	-136	-4,962	-5,409	-1,863

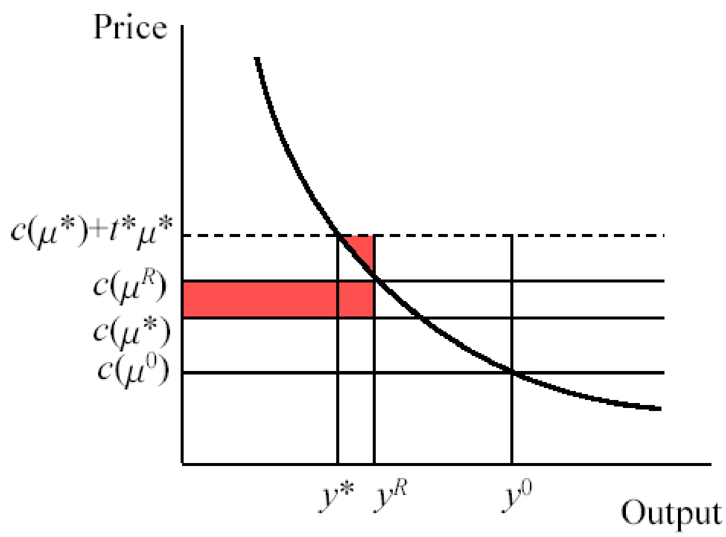
Figure Captions

Figure 1: Partial Equilibrium Efficiency Loss from OBA

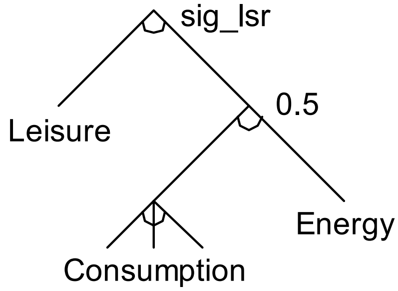
Figure 2: Household utility function

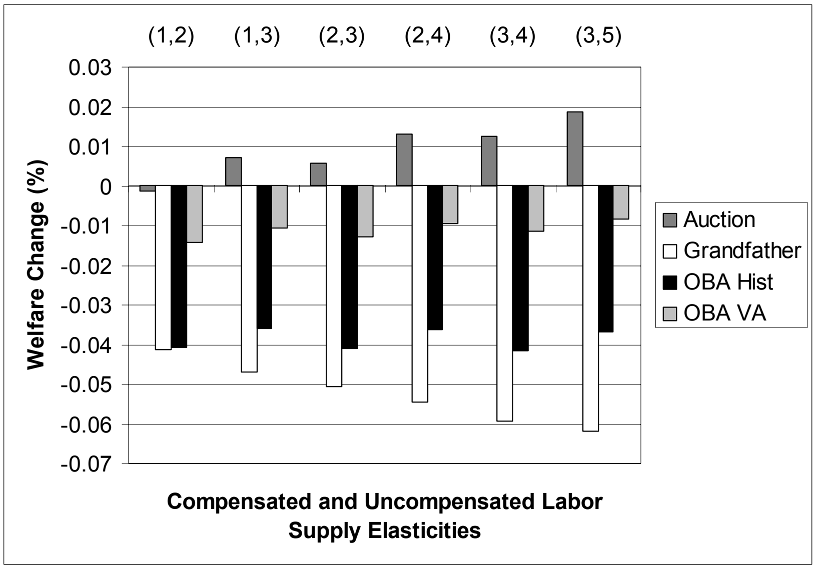
Figure 3: Sensitivity to Labor Supply Elasticities

Figure 4: Sensitivity to Target Stringency

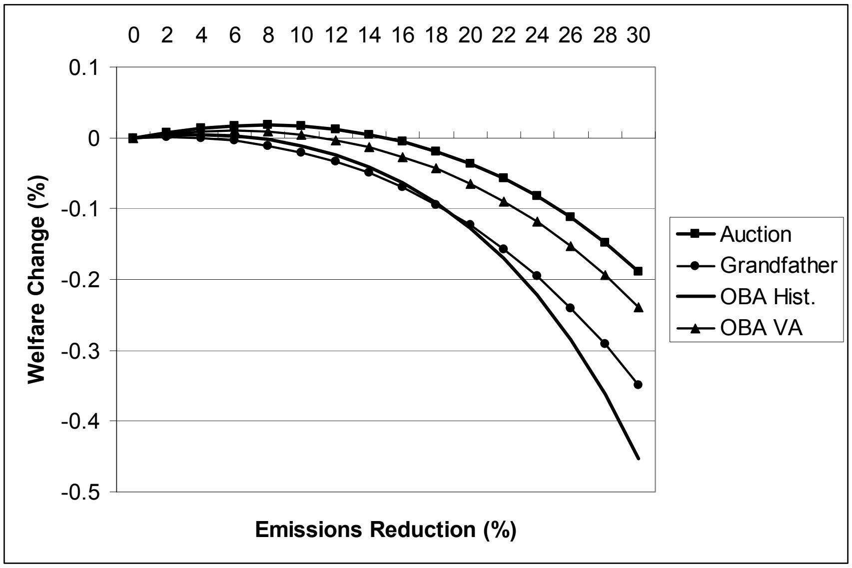


Price
$c(\mu^*)+t^*\mu^*$
$c(\mu^R)$
$c(\mu^*)$
$c(\mu^0)$
y^*
y^R
y^0
Output


sig_lsr
Leisure
0.5
Energy
Consumption



(1,2)
(1,3)
(2,3)
(2,4)
(3,4)
(3,5)
0.03
0.02
0.01
0
-0.01
-0.02
-0.03
-0.04
-0.05
-0.06
-0.07
Welfare Change (%)
Auction
Grandfather
OBA Hist
OBA VA
Compensated and Uncompensated Labor Supply Elasticities



0 2 4 6 8 10 12 14 16 18 20 22 24 26 28 30
0.1
0
-0.1
-0.2
-0.3
-0.4
-0.5
Welfare Change (%)
Emissions Reduction (%)
Auction
Grandfather
OBA Hist.
OBA VA

ENDNOTES

[i] See Burtraw et al. (2002) and Bovenberg and Goulder (2001).

[ii] For a comprehensive collection of the tax interaction literature, see Goulder (2002).

[iii] See e.g. Parry (1996) and Goulder et al. (1997). Fullerton and Metcalf (2001) make the distinction that policies that create scarcity rents (as opposed to policies that raise no revenue) are those that interact with labor tax distortions.

[iv] Parry and Williams (1999) also consider performance standards.

[v] This is the method of Dissou (forthcoming) and Goulder et al. (1999).

[vi] In other words, if the firm loses the permits if it exits the industry, the allocation becomes an operation subsidy that lowers long-run average costs by *tA/y* per unit. See also Boehringer et al. (2002).

[vii] See Fischer (2003).

[viii] We report impacts on 18 nonenergy sectors as well as 5 energy sectors. Most of the major climate models are much more highly aggregated, with 5 or fewer nonenergy sectors (Fischer and Morgenstern 2003). Some have more detail in modeling specific energy supplies. However, there are climate models based on GTAP, such as recent versions of ABARE-GTEM, that can offer richness in all of these dimensions.

[ix] The implicit assumption is that capital reallocates itself more quickly than production functions change. For example, given a real depreciation rate of 5% and an economy-wide real growth rate of 3%, then stopping investment in a sector allows it to shrink by 8% a year relative to the economy. The modest changes in production we see can then be accomplished within a couple years.

[x] Tax data are an area targeted for improvement in GTAP (Babiker et al. 2001).

[xi] See Fox (2002) for the full derivation.

[xii] After that period, emissions are to be further reduced to 1990 levels, though not below, as specified in the Kyoto Protocol targets.

[xiii] These are electric generation, industrial production, commercial activities, and transportation.

[xiv] For instance, including the EU emissions trading program would effectively change the baseline against which we evaluate the U.S. policies. The main relative change would regard emissions leakage to the EU, since emissions are theoretically capped for certain sectors there.

[xv] We did conduct some experiments in which the government budget was held constant through a lump-sum tax. This is the equivalent of taking back a portion of the lump-sum-rebated permit revenue. For the gratis scenarios, the difference was very small, since permit revenues are small and possibly offset by labor supply reactions. Of course, an Auction with lump-sum tax becomes equivalent to grandfathering.

[xvi] This assumption runs somewhat counter to the CSA, which allocates many of the transport sector permits to the upstream energy suppliers.

[xvii] Crude oil does not embody permit costs; those requirements are revealed in the refined oil prices.

[xviii] Production is measured using Laspeyre's volume index, in which changes are valued at *ex ante* prices, so they do not reflect price changes but actual output changes.

[xix]For example, Paltsev et al. (2003) find a welfare cost of 0.05% in 2010 for the McCain-Lieberman CSA Phase I target, with no banking, albeit with the MIT-EPPA model, which is also based on GTAP. EIA (2003) find a decline in GDP of 0.6% in the longer run, by 2025. Over that same timeframe, Smith (2004) finds a GDP loss of 0.7% with grandfathering and 0.4% with enough auctioning to offset revenue changes; however, they project larger

welfare losses than they other studies. Smith et al. (2003) explain that these larger impacts are driven by key intertemporal optimization assumptions in their model (which are absent in this one); when they remove perfect foresight and the ability of consumers to adjust consumption over time, they find consumption losses of less than 0.06%.

[xx] For this reason, grandfathering permits with a lump-sum tax adjustment fares slightly better than this scenario with the labor tax adjustment.

[xxi] For this reason, there would be little difference between lump-sum and labor tax revenue adjustments.

[xxii] This question is a general equilibrium variation of that posed by Bovenberg and Goulder (2001), who calculated the gratis permit shares needed to hold industry profits harmless.

[xxiii] For further discussion, see Fischer et al. (2003).

[xxiv] The European Union has its own "state aid" rules, and the European Commission, in monitoring the national allocation plans, seems to be frowning on explicit updating schemes; however, most plans have aspects of gratis allocation that are not truly lump sum, being conditional on production, and expectations for the second commitment period allocations that create expectations similar to OBA incentives. In the United States, OBA is explicitly allowed—even encouraged—in the formulation of state allocation plans for NO_x trading in the Northeast.

[xxv] Preliminary estimates by Marshall and Hicks (2005) place these costs at $157 billion.

[xxvi] Total is the total value of output, which includes intermediate goods, unlike final demand or GDP.

Ann Wolverton's Comments on the Fisher and Fox paper and the Burtraw, Kahn, and Palmer paper

Prepared for the Proceeding for the Market Mechanism and Incentives Workshop, October 17-18, 2006

- Both papers examine issues relevant to the design of cap-and-trade programs.

    - Fischer and Fox paper focuses on how the permits are allocated and trade-offs between efficiency and equity including different ways to update based on output or value-added

    - Burtraw et al. paper looks at how a safety valve could be used to increase the efficiency of cap-and-trade programs through dynamic adjustment to cost information revealed by permit prices.

- Fisher and Fox - Allocation Mechanisms

    - Auction – can offset other distortions with revenue
    - Grandfathering – cannot offset other distortions with revenue; firms see windfall profits due to value of the allocated permits. This profit is not passed on to consumers in the form of lower electricity prices.
    - Output-based – reallocation based on decision variable. This reallocation introduces inefficiencies into the market by expanding output but distributes rents to consumers as a decrease in prices.
    - Value-added based – less inefficiency because broad based distribution of rents, lower price for consumers

- Comments on Fisher and Fox:

    - Potentially larger rents – risk of political jockeying; rent seeking reallocation of permits that can lead to policy distortions particularly when stakes are large → does this differ across different methods?
    - Perhaps related to this is the question of what happens when assumption of perfect competition is relaxed (at least for industries with a few big industry leaders)
    - A more explicit comparison with performance standards would be useful. How do output-based allocations compare to performance standards?
    - How sensitive are the results to assumptions, e.g, labor-leisure elasticity of substitution; other elasticities; labor tax.
    - What if the U.S. joined in multi-lateral cooperation? Does it change the relative ranking? Does it change the distance between them when rated in terms of efficiency?
    - Also related to the Burtraw et al paper, how does a hybrid cap-and-trade rank? Are the effects similar? Do different incentives for technology change lead to a decrease in marginal costs?

- In terms of regulatory stringency, some estimates predict that greenhouse gas emission will need to decrease by one-half over time to stabilize atmospheric concentrations. At higher reductions, do changes in welfare continue to widen (across the options)? What are the driving factors?

- Comments on Burtraw, Kahn and Palmer:

  - What is the trade-off between policy certainty (for firms) and getting it right through adjustment over time?
  - How important is it for firms to know the role of government? And the rules of the game? (versus modifying the rules over time – firms may then hedge against banked allowances losing value over time). The adjusted rule would have to be estimated beforehand to minimize uncertainty → could get the rule wrong.
  - Interaction between safety valve and banking seems important to evaluate. More generally, behavioral effects/responses from other firms may change the analysis (such as effects on innovation that may decrease marginal costs).
  - How do other alternative mechanisms for dealing with uncertainty compare? (e.g., incentives for development of alternative fuels → RFS, subsidies, etc.?)
  - Does it create perverse incentives for government? Some of the problems with RECLAIM could have been averted with more flexibility in design (such as banking). With safety valves, there may not be an incentive to pay as close attention to design since you have a ready out.
  - Looking at the case where the marginal benefit curve is not flat seems like an interesting extension (cases other than CO_2)

- What are the interaction/synergies between the mechanisms discussed in the two papers? It is possible that updating, which increases the inefficiency some amount, could be used as an option to re-evaluate whether the number of permits was correctly allocated based on cost information revealed by the market → adjustment mechanism incorporated?

Arik Levinson's Discussant Comments on Fischer and Fox and Burtraw, Kahn, and Palmer

Fischer and Fox "Optimal Output-Based Allocation of Emissions Permits for Mitigating Tax and Trade Interactions"

<u>Intro</u>

- Fischer/Fox, good economist/contrarians, put forth good argument for case where trading is <u>not</u> necessarily, second best (or even third best).

- Long economic history picking on bad govt. policies various govts have enacted (makes for entertaining analysis). Fischer/Fox have moved on -- now picking on bad policies govts might be thinking of someday adopting: periodically updating the allocation of emissions permits based on firms' output. Not the first to model this type of thing.
  - Also F/F make nice point. Grandfathering itself has an element of OBA in it, in that it is a subsidy for continued operation of firms that might otherwise go out of business, if they'd lose their allocation by doing so.
  - Oates & Schwab 1988, Heutel and Fullerton 2006.

- This paper succeeds at convincing me of several things I would not have guessed before I started reading the paper:
  - Can it be optimal to adjust permit allocations among firms based on their output? (In second best world, yes.)
  - If tradable permits are handed out among various sectors, should we allow cross-sector trade?

- Great paper. Presents partial eq'm model, which has much of the intuition. We know it's wrong, but we can see all the moving parts and know which parts of the intution are likely to generalize and which way the PE biases might work. Then presents CGE. We know that's wrong too, but have no idea why.

<u>Issues</u>

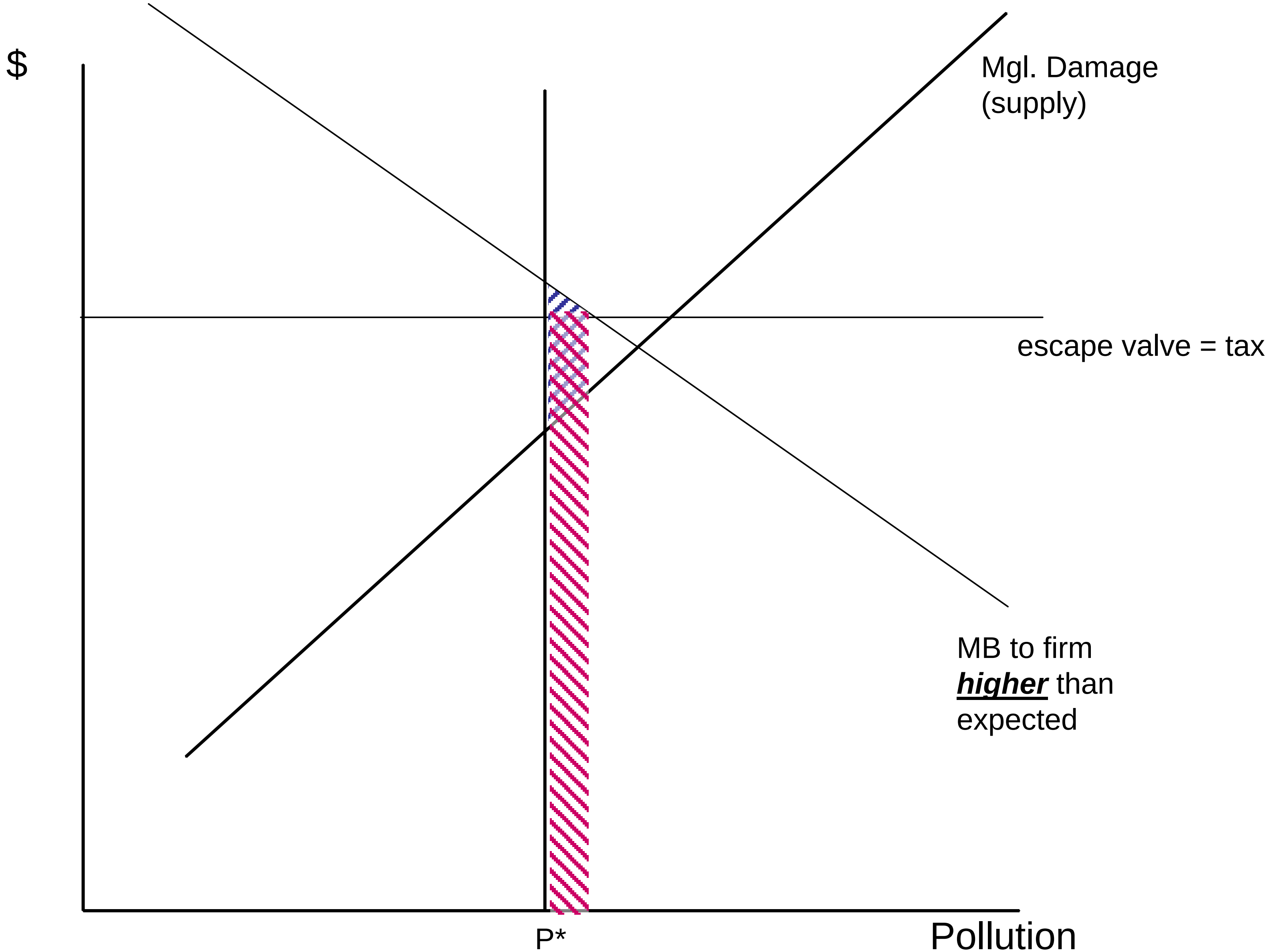
- Paper analyzes an odd policy. First we hand out permits, by sector, based on their historical *emissions* (or *value added*). Then we allow trading of emissions permits. Then we periodically update the permit allocations based on *output*.
  - Why not allocate to firms in the first place, rather than by sector. Then there's no issue about whether or not to allow inter-sectoral trade?
- First best = auction permits.
  - Problems.
    - Political feasibility

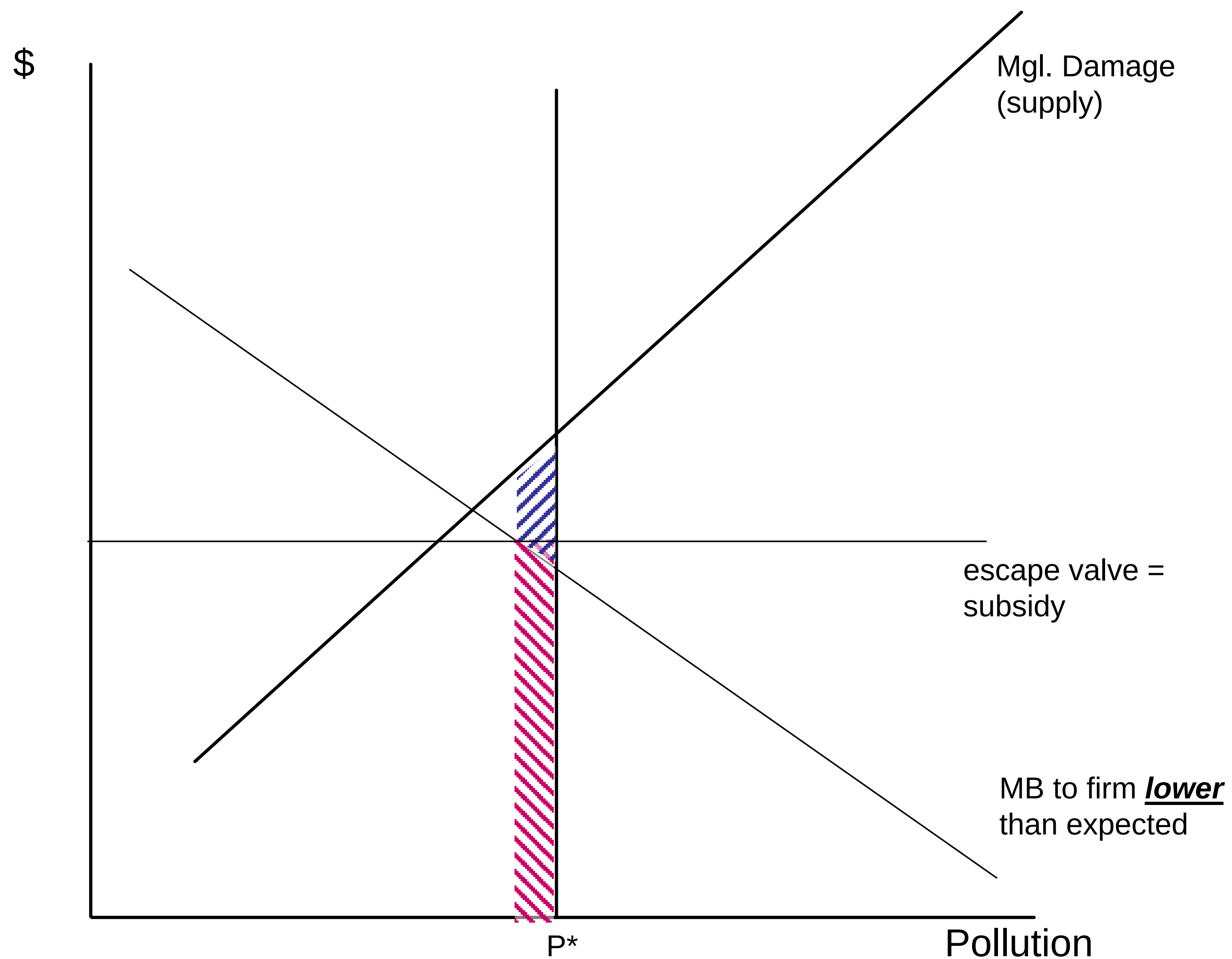
        - Tax interaction effects in 2nd best world. (Raises cost of goods, lowering real wage, increases DWL from labor tax.)
        - Leakage. (High price of goods encourages substitution for sectors or countries not regulated.)
- Second best = give away the permits to firms.
    - Among methods to give away: grandfather (lump sum) or adjust (OBA). I would have thought grandfather=2nd best and OBA=3rd best.
        - grandfather solves political feasibility, but not tax interaction/leakage.
        - OBA subsidizes output, lowers mkt price of good, reduces tax interation and leakage
            - but raises mgl abatement cost (we're not abating in least-cost way -- by reducing output), raises production cost.

Notes for Fischer/Fox

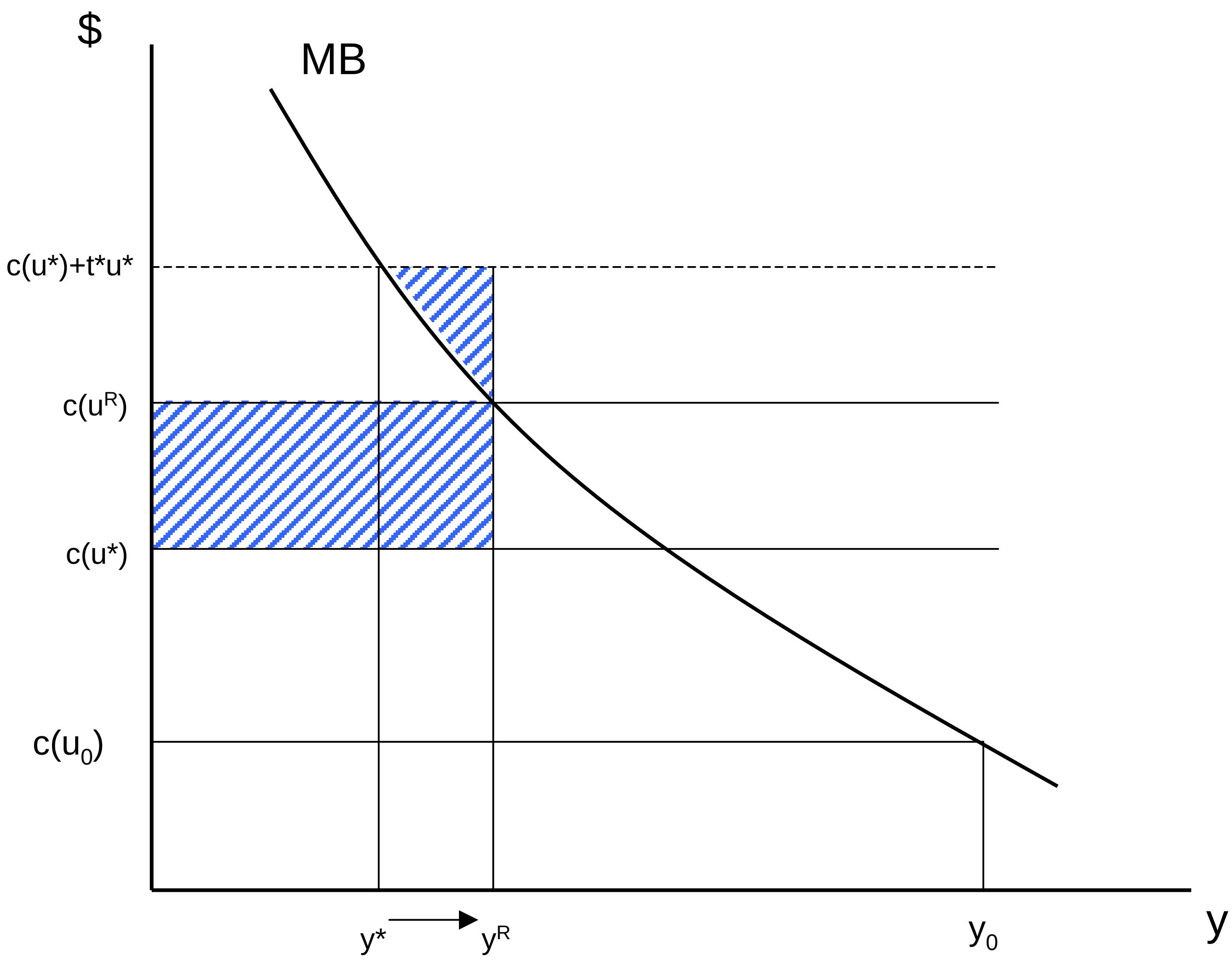
- Sensitivity analysis. Do table 3 (or 12 or 13) under a variety of scenarios:
    - change assumptions about labor elasticity or hours in the day, etc. tax rates, etc.
    - (More of figures 3 and 4.)
- p. 6. Define "performance standards" earlier, and more clearly.
- Why would you hand out permits by sector? I.e. output based or VA based or historical emissions based permits could be handed out (and adjusted) on a firm-by-firm basis. Or define the whole economy as one sector.
- Figure 1 took me some time to figure out. The paragraph on p. 12 is pretty cryptic. Why the first part of DWL" the higher-than optimal production costs"? Because firms have produced more, and therefore need to employ more costly abatement techniques in order to reduce their emissions rates? Doesn't that come with an associated benefit, in the form of cleaner air? That should lower the externality and reduce that line, but I see after some puzzling that total damage is the same at both output levels, because total emissions are capped.
- p.12, "It is worth noting that applying separate cap and trade programs with output-based allocations to multiple sectors is equivalent to setting performance standards." Why? By "performance standards" do you mean ratios of emissions per unit output?
- Why is thinking about trading across sectors in a multisector market different than thinking about trading across firms within a sector? Firms within a sector have different cost structures, emissions, and demand elasticities (all the things that make trading cost-effective). I guess this is the same as asking why not just have overall allocation by historical emissions or value added, then allow trade. I.e. just define the whole economy as one sector.
- Top of p. 14, say explicitly whether sector 1 buys or sells permits. (I confess it took me more than a minute to figure out.)
- Other papers that model emissions permits based on outputs (or inputs)
    - Oates and Schwab JPubE (1988) pollution allowed (non-tradable) based on labor.
    - Fullerton and Heutel (2005). "The General Equilibrium Incidence of Environmental Mandates" (Study incidence of regulation that affects emissions per unit of output in Harberger Framework.)
- The differences in table 12 seem extraordinarily small. (<3% of the damages from one hurricane, as you note). They seem well within any errors in the CGE model.
    - That said, the amounts (as opposed to percentages) are significant. Is it truly the case that OBE will always dominate grandfathering, or can you input parameters where the orderings in figure change. (Looks like it does from the undiscussed figures 3 and 4.

Burtraw, Kahn, and Palmer

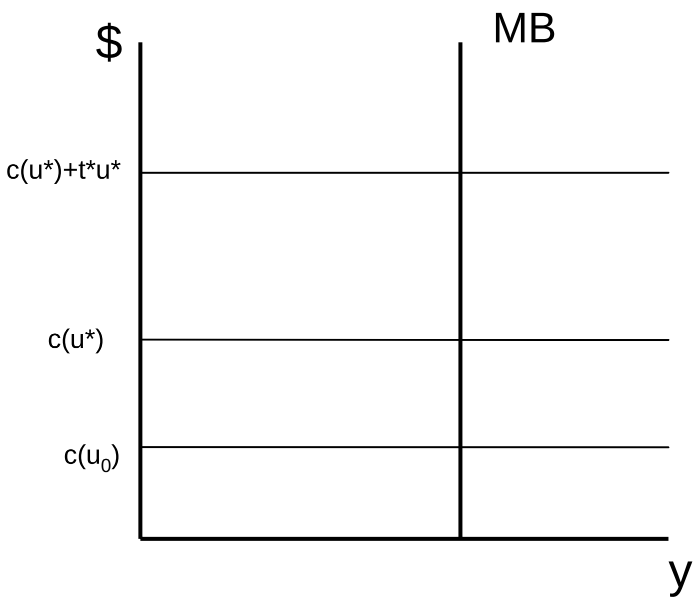



$
Mgl. Damage
(supply)
escape valve =
subsidy
MB to firm ***lower***
than expected
P*
Pollution

Fischer and Fox -- Figure 1

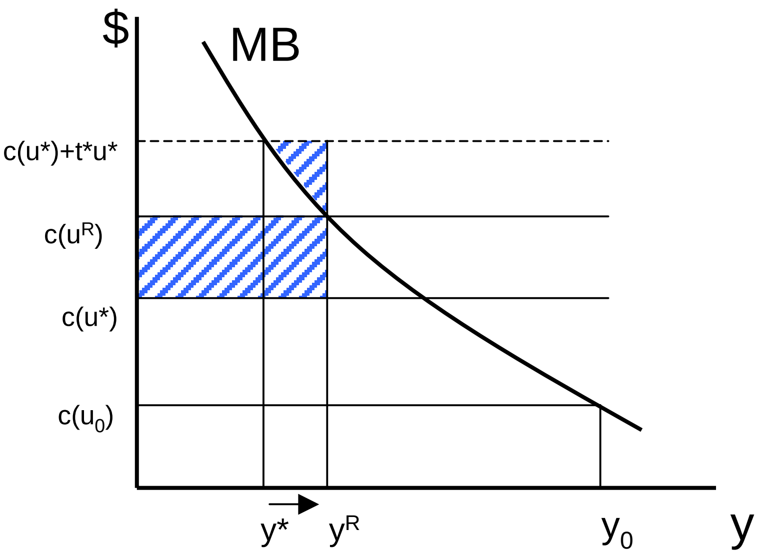


Sector 1



- consumers get 100% output subsidy
- no efficiency loss
- $u^R = u^*$
- $c(u^R) = c(u^*)$

Sector 2



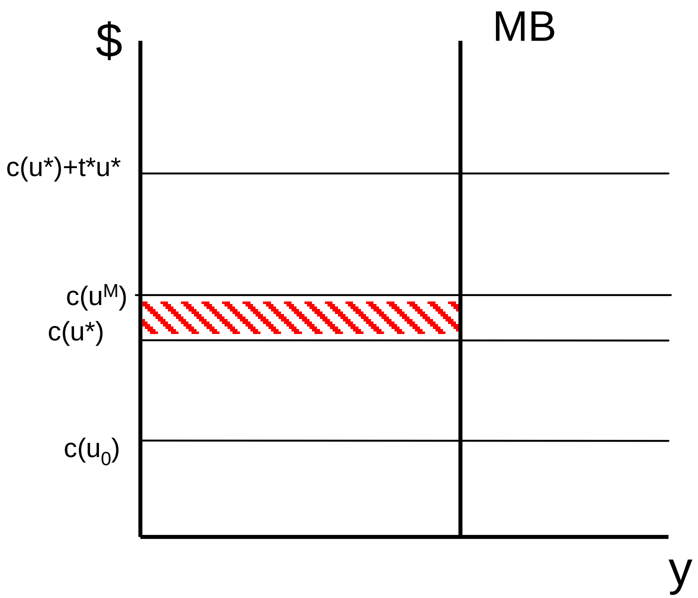
- output subsidy
- efficiency loss
- $u^R < u^*$
- $c(u^R) > c(u^*)$

$$\boxed{c'(u_1) < c'(u_2)}$$

Allow Trading Across Sectors

Sector 1

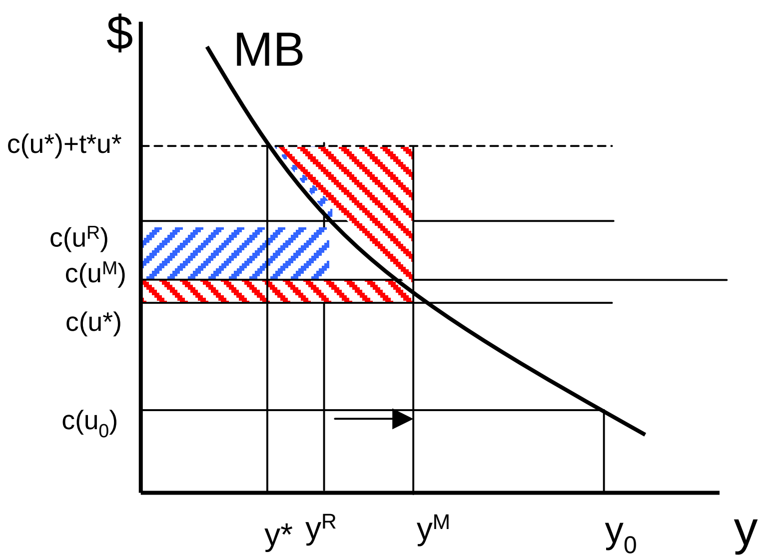
•sells permits, passing compliance cost to customers



•consumers get less of the output subsidy
•Sector 1 shares some of the efficiency loss
•$u^M < u^*$
•$c(u^M) > c(u^*)$

Sector 2

•buys permits, to reduce compliance cost of extra output



•increased output subsidy
•efficiency loss changes
•$u^R < u^R < u^*$
•$c(u^R) > c(u^R) > c(u^*)$

Question before turning to CGE

• Relabel "sectors" as "firms" and we're back to a case against trading in the first place.

• We might as well partition the market by region, or alphabetically.

• Why the arbitrary division of the permit market into sectors?

  - McCain-Lieberman Act?
  - GTAP model constraints?

• Amounts to an diminution of the gains from trading permits.

The CGE Model

GTAP + energy (GTAP-EG) + carbon + labor/leisure

Many assumptions.

- labor income tax rates 40% and 20%
- uncompensated labor supply elasticity 0.1
- benchmark elasticity of subst. between labor and leisure 1.736 in US
- policy goal reduce CO2 by 14%
- no caps in other countries (maximize leakage)
- energy CES nested to three levels
- ...
- ...
- ...
- ...

Table 12: Summary Indicators for the United States

Indicator	*Unit*	*Auction*	*Grandfather*	*Hist. OBA*	*VA OBA*
Welfare	% change in equivalent variation	0.00	-0.05	-0.04	-0.01
Production	% change	-0.38	-0.54	-0.36	-0.42
Employment	% change	0.12	-0.15	0.00	0.03
Real wage	% change	0.46	-0.59	-0.02	0.09
Labor tax change	percentage pts	-1.81	0.00	-0.34	-0.40
Carbon leakage	% of reductions	15.4	15.3	12.5	15.3
Permit price	$/metric ton C	$49.21	$48.59	$70.47	$49.87

1. VA OBA *is* auctioning.
2. Differences small, given uncertainties in CGE.

Insights

- OBA not necessarily worse that grandfathering
- Trading among sectors not necessarily more efficient.

Market Mechanisms and Incentives: Applications to Environmental Policy

A Workshop Sponsored by the U.S. Environmental Protection Agency's National Center for Environmental Economics (NCEE) and National Center for Environmental Research (NCER)

Resources for the Future
1616 P Street, NW, Washington, DC 20036
October 17-18, 2006

Market Mechanisms and Incentives: Applications to Environmental Policy

Resources for the Future
1616 P Street, NW, Washington, DC 20036
(202) 328-5000

October 17th – 18th, 2006

October 17, 2006: Market Mechanisms in Environmental Policy

8:00 a.m. – 8:45 a.m. **Registration**

8:45 a.m. – 11:45 a.m. **Session I: Brownfields and Land Issues**
Session Moderator: **Robin Jenkins**, EPA, National Center for Environmental Economics

8:45 a.m. – 9:00 a.m. Introductory Remarks: **Sven-Erik Kaiser**, EPA, Office of Brownfields Cleanup and Redevelopment

9:00 a.m. – 9:30 a.m. Environmental Liability and Redevelopment of Old Industrial Land
Hilary Sigman, Rutgers University

9:30 a.m. – 10:00 a.m. Incentives for Brownfield Redevelopment: Model and Simulation
Peter Schwarz and **Alex Hanning**, University of North Carolina at Charlotte

10:00 a.m. – 10:15 a.m. **Break**

10:15 a.m. – 10:45 a.m. Brownfield Redevelopment Under the Threat of Bankruptcy
Joel Corona, EPA, Office of Water, and **Kathleen Segerson**, University of Connecticut

10:45 a.m. – 11:00 a.m. Discussant: **David Simpson**, EPA, National Center for Environmental Economics

11:00 a.m. – 11:15 a.m. Discussant: **Anna Alberini**, University of Maryland

11:15 a.m. – 11:45 a.m. Questions and Discussion

11:45 a.m. – 12:45 p.m. **Lunch**

12:45 p.m. – 2:45 p.m. **Session II: New Designs for Incentive-Based Mechanisms for Controlling Air Pollution**
Session Moderator: **Will Wheeler**, EPA, National Center for Economic Research

12:45 p.m. – 1:15 p.m. Dynamic Adjustment to Incentive-Based Environmental Policy To Improve Efficiency and Performance
Dallas Burtraw, Danny Kahn, and Karen Palmer, Resources for the Future

	1:15 p.m. – 1:45 p.m.	Output-Based Allocation of Emissions Permits for Mitigating Tax and Trade Interactions **Carolyn Fischer**, Resources for the Future
	1:45 p.m. – 2:00 p.m.	Discussant: **Ann Wolverton**, EPA, National Center for Environmental Economics
	2:00 p.m. – 2:15 p.m.	Discussant: **Arik Levinson**, Georgetown University
	2:15 p.m. – 2:45 p.m.	Questions and Discussion
2:45 p.m. – 3:00 p.m.	**Break**	
3:00 p.m. – 5:30 p.m.	**Session III: Mobile Sources** Session Moderator: **Elizabeth Kopits**, EPA, National Center for Environmental Economics	
	3:00 p.m. – 3:30 p.m.	Tradable Fuel Economy Credits: Competition and Oligopoly **Jonathan Rubin**, University of Maine; **Paul Leiby**, Environmental Sciences Division, Oak Ridge National Laboratory; and **David Greene**, Oak Ridge National Laboratory
	3:30 p.m. – 4:00 p.m.	Do Eco-Communication Strategies Reduce Energy Use and Emissions from Light Duty Vehicles? **Mario Teisl**, **Jonathan Rubin**, and **Caroline L. Noblet**, University of Maine
	4:00 p.m. – 4:30 p.m.	Vehicle Choices, Miles Driven, and Pollution Policies **Don Fullerton**, **Ye Feng**, and **Li Gan**, University of Texas at Austin
	4:30 p.m. – 4:45 p.m.	Discussant: **Ed Coe**, EPA, Office of Transportation and Air Quality
	4:45 p.m. – 5:00 p.m.	Discussant: **Winston Harrington**, Resources for the Future
	5:00 p.m. – 5:30 p.m.	Questions and Discussion
5:30 p.m.	**Adjournment**	

October 18, 2006:

8:45 a.m. – 9:15 a.m.	**Registration**	
9:15 a.m. – 12:20 p.m.	**Session IV: Air Issues** Session Moderator: **Elaine Frey**, EPA, National Center for Environmental Economics	
	9:15 a.m. – 9:45 a.m.	Testing for Dynamic Efficiency of the Sulfur Dioxide Allowance Market **Gloria Helfand**, **Michael Moore**, and **Yimin Liu**, University of Michigan
	9:45 a.m. – 10:05 a.m.	When To Pollute, When To Abate: Evidence on Intertemporal Use of Pollution Permits in the Los Angeles NOx Market **Michael Moore** and **Stephen P. Holland**, University of Michigan

10:05 a.m. – 10:20 a.m.	**Break**	
	10:20 a.m. – 10:50 a.m.	A Spatial Analysis of the Consequences of the SO_2 Trading Program **Ron Shadbegian**, University of Massachusetts at Dartmouth; Wayne Gray, Clark University; and Cynthia Morgan, EPA
	10:50 a.m. – 11:20 a.m.	Emissions Trading, Electricity Industry Restructuring, and Investment in Pollution Abatement **Meredith Fowlie**, University of Michigan
	11:20 a.m. – 11:35 a.m.	Discussant: **Sam Napolitano**, EPA, Clean Air Markets Division
	11:35 a.m. – 11:50 a.m.	Discussant: **Nat Keohane**, Yale University
	11:50 a.m. – 12:20 p.m.	Questions and Discussion
12:20 p.m. – 1:30 p.m.	**Lunch**	
1:30 p.m. – 4:35 p.m.	**Session V: Water Issues** Session Moderator: **Cynthia Morgan**, EPA, National Center for Environmental Economics	
	1:30 p.m. – 2:00 p.m.	An Experimental Exploration of Voluntary Mechanisms to Reduce Non-Point Source Water Pollution With a Background Threat of Regulation Jordan Suter, Cornell University, Kathleen Segerson, University of Connecticut, Christian Vossler, University of Tennessee, and Greg Poe, Cornell University
	2:00 p.m. – 2:30 p.m.	Choice Experiments to Assess Farmers' Willingness to Participate in a Water Quality Trading Market **Jeff Peterson**, Washington State University, and Sean Fox, John Leatherman, and Craig Smith, Kansas State University
2:30 p.m. – 2:45 p.m.	**Break**	
	2:45 p.m. – 3:15 p.m.	Incorporating Wetlands in Water Quality Trading Programs: Economic and Ecological Considerations **Hale Thurston** and Matthew Heberling, EPA, National Risk Management Research Laboratory, Cincinnati, Ohio
	3:15 p.m. – 3:35 p.m.	Designing Incentives for Private Maintenance and Restoration of Coastal Wetlands **Richard Kazmierczak** and **Walter Keithly**, Louisiana State University at Baton Rouge
	3:35 p.m. – 3:50 p.m.	Discussant: **Marc Ribaudo**, USDA, Economic Research Service
	3:50 p.m. – 4:05 p.m.	Discussant: **Jim Shortle**, Pennsylvania State University
	4:05 p.m. – 4:35 p.m.	Questions and Discussions
4:35 p.m. – 4:45 p.m.	**Final Remarks**	

4:45 p.m. **Adjournment**

Tradable Fuel Economy Credits:
Competition and Oligopoly[1]

By

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Abstract

Corporate average fuel efficiency (CAFE) regulations specify minimum standards for fuel efficiency that vehicle manufacturers must meet independently. We design a system of tradeable fuel economy credits that allows trading across vehicle types and manufacturers with and without considering market power in the credit market. We perform numerical simulations to measure the potential costs savings from moving from the current CAFE system to one with stricter standards, but that allows vehicle manufacturers various levels of increased flexibility. We find that the ability for each manufacturer to average credits between its cars and trucks provides greater savings than the ability to trade credits across manufacturers in separate vehicle markets. As expected, the greatest savings comes from the greatest flexibility in the credit system. Market power lowers the potential cost savings to the industry as a whole. However, loss in efficiency from market power does not eliminate the gains from credit trading.

Key Words: GHG, Credits, Cost-Benefit, Socio-Economic, Energy Conservation

JEL codes: Q25, Q28, Q30, Q48, Q40

1 Introduction

1.1 Fuel Economy Standard Policy Context

Corporate average fuel efficiency (CAFE) standards established by the US Energy Policy and Conservation Act of 1975 (PL94-163) specify minimum fleet average standards for fuel efficiency that U.S. light-duty vehicle (car and light-truck) manufacturers must meet. Light-duty vehicles produced 59% of transportation CO_2 emissions in 2003 (USEPA, 2005, p. 57) and consume 36% of the oil used in the U.S. (Davis and Diegel, 2004, Tables 1.13, 2.3, 2.4.)

The effectiveness of CAFE standards in raising the light-duty vehicle fleet's fuel efficiency, and other effects of CAFE regulations, have been discussed in a large body of literature. It was debated whether the improvements in average fuel efficiency realized from 1978 (the first year that the CAFE standards went into effect) through 1987 were attained at a reasonable economic cost and whether the CAFE regulations induced undesirable changes in vehicles that could lower their safety (Greene (1990), Crandall and Graham (1989), Nivola and Crandall (1995), Greene (1998)).

Thorpe (1997) found that the CAFE standards have led to a shift toward larger, more luxurious models in the imported Asian fleet and may have led to a decrease in the fleet's average fuel efficiency. In addition, the CAFE standards themselves, by being less restrictive for trucks than for cars, may have had the unintended effect of encouraging the shift in market share from cars to light-duty trucks. The light-duty truck share of new vehicle sales has grown from 9.8% in 1979 to 42.8% in 1997 (Godek, 1997; NHTSA, 1998a, p. 16627). Parry, et al. (2004) examine the social welfare of raising CAFE standards taking into consideration existing externalities. They find that higher CAFE standards can produce anything from moderate welfare gains to substantial welfare losses, depending on how consumers value fuel economy technologies and their opportunity costs.

In 2002 the National Research Council's comprehensive review of the effectiveness and impact of CAFE standards concluded that while the CAFE program has clearly increased fuel economy, certain aspects of the CAFE program have not functioned as intended. These include indirect consumer and safety costs, a breakdown in the distinctions between foreign and domestic fleets, and between minivans, SUVs and cars in the calculation of fuel economy standards, and the artificial creation of fuel economy credits for multi-fuel vehicles.[2] Moreover, the National Research Council concluded that technologies exist that, if applied to light-duty vehicles, would significantly reduce fuel consumption within 15 years (Finding 5).

The availability of improved technologies for fuel economy alone is not sufficient to encourage their widespread adoption. The National Research Council concluded that raising the CAFE standard would reduce future fuel consumption, but that other policies could accomplish this same end at lower cost and greater flexibility. The National Research Council concluded (Finding 11): "Changing the current CAFE system to one featuring tradable fuel economy credits and a cap on the price of these credits appears to be particularly attractive. It would provide incentives for all manufacturers, including those that exceed the fuel economy targets, to continually increase fuel economy, while allowing manufacturers flexibility to meet consumer preferences."[3]

We investigate the potential cost savings from the implementation of a system of tradable fuel economy credits coupled with higher fuel economy standards. These benefits include the economic cost savings from reduced fuel use, reductions in fuel use and reductions in US GHG emissions from the light-duty vehicle sector.

1.2 Current CAFE Regulations and Standards

Current legislation and regulation requires that each manufacturer of passenger cars or light trucks with a gross vehicle weight rating of 8,500 lbs. (3636.4 Kg) or less manufactured for sale in the US

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[2]The majority of members of the Committee on the Effectiveness and Impact of CAFE standards found that down-weighting and down-sizing, in part due to CAFE standards, increased traffic fatalities. Dissenting minority committee members, including David Greene, concluded that the statistical evidence for such safety effects is not conclusive.

[3] Similar views are also expressed by the the National Commission on Energy Policy (NCEP, 2004) and the Pew Center on Global Climate Change (2006).

attain a minimum corporate average fuel efficiency standard (PL 94-163, 49 U.S.C. §32902). Regulations adopted in March, 2006 change the structure of the corporate average fuel economy for light trucks and establishes higher CAFE standards for model year 2008-2011 light trucks (49 CFR Parts 523, 533 and 537). Starting in MY 2011, the CAFE program will include trucks that have a gross vehicle weight up to 10,000 lbs (NHTSA, 2006, p. 17).[4]

The CAFE standard for each manufacturer, *m*, is defined as the sales-weighted harmonic average fuel economy, defined in terms of miles per gallon: E^*_{vo} (49 U.S.C. §32902, §32904). There are separate standards for each vehicle class *v* (passenger car or light truck) and origin of manufacturing for cars, *o* (domestic or foreign).[5] Thus, S_{mvi} are manufacturer *m's* sales in vehicle class *v*, all models *I*. The form of the harmonic average standard is given below. [6]

$$E^*_v \leq \frac{S_{mv}}{\sum_i \frac{S_{mvi}}{E_{mv}}}, \quad where \quad S_{mv} = \sum_i S_{mvi} \tag{1}$$

If a manufacturer does not meet the standard, it is liable for a civil penalty of $5.5 for each 0.1 mile per gallon (or $55/MPG) its fleet average falls below the standard, multiplied by the number of vehicles it sold in a given model year in each fleet. Credits are earned when a manufacturer more than attains the standard in any model year. These credits may be carried forward (banked) or carried back (borrowed) for

[4]There are additional other specific rules and guidelines given in the final rule. NHTSA estimates that expanding the truck category will add an additional 240,000 vehicles into the CAFE program in 2011.

[5]For simplicity we do not separate out foreign and domestic car fleets.

[6]New CAFE regulations for light-duty trucks due to be phased in are based on a measure of vehicle size called "footprint," the product of multiplying a vehicle's wheelbase by its track width. The form of the standard is as given in the formula above, except that the standard S_v or target T, for trucks is given as: $T = \frac{1}{\frac{1}{a} + \left(\frac{1}{b} + \frac{1}{a}\right)\frac{e^{(x-c)/d}}{1+e^{(x-c)d}}}$ where a, b, c and d are parameters representing maximum and minimum fuel economy targets, footprint values and rates of change targets (NHTSA, 2006, p. 178).

three years on a rolling basis. Important limitations of the current system are that fuel economy credits are not tradable amongst manufacturers nor subclassification for a given manufacturer.[7]

Level of current standards and proposed regulations

The current level of fuel economy standard for passenger cars is 27.5 mpg. The standard is set at 21.0 mpg for light-trucks produced through MY 2005. This rises to 22.2 mpg for MY 2007 (Federal Register, 49 CFR Part 533), and the new standards promulgated by NHTSA raise the standard for light-trucks to 24 mpg by MY 2011 and allow compliance based upon a reformed CAFE standard based on a manufacturer's vehicles footprint.

In their report, the National Research Council determined that the cost-effective average fuel economy could be increased by 12% for subcompact automobiles, up to 27% for large passenger cars and between 25% and 42% for light-duty trucks (depending on size) over the next 15 years (National Research Council, 2002; p. 66).[8] Given these benchmarks, we examine two alternative fuel economy levels, 30% and 40% improvements by 2015. Given a base year fuel economy standard of 27.5 for passenger cars, 30% and 40% improvements implies targets of 35.75 and 38.5 MPG, respectively. The corresponding targets for light trucks are 26.9 and 29.0 (compared to a base level of 20.7), and the combined light-duty fleet numbers are 31.4 and 33.8, versus a 2002 model year weighted average of 24.2. Note that these targets are relative to base year fuel economy standards, (using the MY light truck share of 48.9%) not the base year fleet fuel economy level actually attained (NHTSA, 2003, Table II-6).

2 Market Models of Producer and Consumer Behavior

[7]An important aspect of the current CAFE system is the value of time flexibility to manufacturers. As shown by Rubin and Kling (1993) in the context of phasing in stricter standards for new vehicles for criteria emissions, a credit system can realize cost savings when firms are allowed to borrow and banking credits even if they do not trade. We examine the value of time flexibility in on-going work.

[8]Cost-effective technologies means combinations of existing and emerging technologies that would result in fuel economy improvements sufficient to cover the purchase price increases they would require holding size, weight, and vehicle performance characteristics constant.

2.1 The Fuel Economy Market Model - Perfect Competition

Given a market for tradable fuel economy credits, we formulate the objective from the perspective of a vehicle manufacturer which maximizes the net private value to consumers of vehicle fuel efficiency plus the revenues from fuel economy credits sold (or purchased) for each vehicle type. The net value of fuel efficiency is the consumer's valuation of vehicle-lifetime fuel savings minus the increase in vehicle cost due to fuel economy technology. We examine two cases of consumer valuation of fuel economy. In our high value case, the representative consumer carefully calculates the value of fuel savings over the full-expected life of the vehicle. Our alternative hypothesis assumes that consumers consider only the first three years of fuel savings but do not discount the savings. In general, this implies that consumers will place less than half as much value on fuel savings. In theory, failing to account for real future fuel savings would represent a market failure, in the sense that real-world consumers would not be acting like the fully informed, rational consumers of economic theory and, thus, the market for fuel economy would not be economically efficient.[9]

Manufacturers could use technology for improving fuel economy to increase performance or to cross-subsidize particular makes and models to alter their distribution of vehicle sales. We expect this latter behavior not to be significant, however, since Greene (1991) has shown that pricing strategies and mix changes are a relatively expensive means for a manufacturer to increase its corporate average MPG.

Other researchers, Parry, et al. (2004) have taken a different approach, one that looks at maximizing social welfare of a representative agent taking into consideration existing externalities (carbon emissions, oil dependency, accidents, and congestion) and preexisting fuel taxes. They find that

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[9]Certainly consumers are heterogenous with differing discount rates and annual vehicle miles of travel. Consumers use their vehicles differently, demand different rates of return, and have different preferences for fuel economy versus other vehicle attributes. To some extent, the differences amongst manufacturers' current fleet fuel economy levels can be explained by the different market segments they serve. No attempt is made here to account for such differences in consumer preferences across manufacturers. For this reason, it is most appropriate to interpret the predicted impacts of alternative standards on manufacturers as being generally indicative of the kinds of impacts the standards may have, rather than as a prediction of the impacts on a particular manufacturer.

raising CAFE could cause significant welfare losses largely (though not exclusively) by lowering the cost per mile driven and exacerbating mileage-related external costs such as congestion, accidents and local pollution. They argue that alternative policies such as broad-based oil and carbon taxes, higher fuel taxes, pay-as-you-drive auto insurance, subsidies for alternative fuel vehicles, and subsidies for R&D into carbon capture technologies are more likely to lead to social welfare improvements.

We agree with Parry et al. that other policies such as higher fuel taxes have desirable efficiency properties.[10] However, we do not agree with their conclusions about the potential negative welfare effects from raising CAFE standards, especially if reformed to allow for additional regulatory flexibility such as credit trading. Where we differ is in looking at the CAFE policy tool in the context of other policy initiatives. Since light-duty vehicle use has many externalities, our view is that this calls for multiple policy tools. In particular, asking fuel efficiency regulations to be responsible for congestion externalities is too much.[11]

2.2 Market Power in Tradable Credits

There are only 15 vehicle manufacturers to whom the fuel economy regulations apply. The top five firms accounted for 82 % of total U.S. sales in 2003, and 84% in MY2004.[12] Moreover, certain fundamentals of the automobile market are not likely to change. Given the economies of scale of automobile production, further consolidation seems more likely than an increase in the number of firms. Given the structure of the CAFE market where credits apply at the manufacturer level, it seems almost

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[10]In addition we have informal evidence from discussion with vehicle manufacturers that consumers want a short, 3-5 year, payback on fuel economy technology. Thus, optimally correcting for this myopia via fuel taxes would require very substantial externality taxes given that new vehicles last about 14 - 16 years.

[11] Consider for example the congestion charging system of the City of London (UK). This congestion charging scheme levies a £8 daily charge for vehicles entering or parking within the city center during peak hours. The Department for Transport estimates that congestion has been reduced by 30%, the number of vehicles entering the zone has been reduced by 18%, air pollution from road traffic in the form of NO_x and particulates have been reduced by 12% and green house gas emissions by 19% since the program took effect in February 2003 (Transport for London, 2004a, p. 1, 2004b, p. 4)

[12]The largest manufacturers, in order of MY2004 sales in the U.S. light-duty vehicle market, were General Motors (4.3 million vehicles), DaimlerChrysler (3.2), Ford (2.9), Toyota (2.1) and Honda (1.3). All 10 other manufacturers sold 2.6 million vehicles (NHTSA, 2005)

inescapable that the market *in tradable credits* will be imperfectly competitive: an oligopoly versus an oligopsony with a competitive fringe.

2.2.1 Incentives for the Exercise of Market Power in Credit Markets

The issue of market power in tradable credit markets has been subject to extensive theoretical and empirical research that includes Hahn (1984), Sartzetakis (1997), Ellerman and Decaux (1998), Misiolek and Elder (1989), Malueg (1990), Innes, et al. (1991), Fershtman and Zeeuw (1995), Westskog (1996 and 2001), and Godby (2002). In these papers, either dominant buyers (monopsony or oligopsony) or sellers (monopoly or oligopoly) may be able to exert market power in the credit market or use their market power in the credit market to gain power in the product market.[13]

In the context of GHG emission credits, Westskog (1996) extends Hahn's (1984) model a monopoly with a competitive fringe to a group of nations as acting as Cournot-players with a competitive fringe. The Cournot players act as leaders deciding how many credits to buy or sell given the other Cournot countries' sales or purchases of credits and given the response function of the followers. The competitive fringe acts as followers who choose the optimal amount of credits to sell or buy given the market price of credits resulting from the first move of the leaders. Similar to Hahn, Westskog finds that the least-cost efficient solution will attain only when the countries with market power are given the number of credits that they want to have after credit trading has taken place.

2.3 Defining CAFE Credits

In much of credit literature, the total sum of credits is set by an environmental regulator. With fuel use credits, however, the total number of credits is determined based on a performance standard set

[13]In addition, firms with market power the credit market may engage in exclusionary manipulation to make gains in the product market (Misiolek and Elder, 1989; Godby, 2002, Innes, et al., 1991). Output market manipulation by vehicle manufacturers who also have market power in a CAFE credit market may be a real possibility. At the corporate nameplate level, this type of manipulation would seem likely. At the same time, however, market power in the vehicle output market is not likely to be maintained at the level of specific makes and models where vehicles compete. Moreover, we do not believe it is practical to characterize accurately market power in the vehicle market, especially as it may relate to reactions to CAFE credit market manipulations. We therefore, do not consider further this potential line of inquiry.

by the NHTSA and the number and sales mix of vehicles chosen by manufacturers. Because the CAFE constraint applies to a harmonic average of MPGs, the exposition is much clearer and the analysis is simplified when the standard is written in terms of fuel intensity (gallons per 100 miles, or GPHM) than fuel economy (miles per gallon).[14] Written in fuel intensity G_{mvi} with the standard (maximum fuel intensity) for vehicle class v denoted by G^*_v, the CAFE regulatory constraint on each manufacturer m is linear:[15]

$$G_{mv} \equiv \sum_i \frac{S_{mvi}}{S_{mv}} G_{mvi} \leq G_v^* \tag{2}$$

The market that will emerge, if credit trading is allowed, is a market for fuel-use credits. Credit quantities will be in units of vehicle-GPHM and credit prices will have units of \$/veh-GPHM.

2.4 Private Market Model: Perfect Competition

Formally, the manufacturer is assumed to maximize (on behalf of the consumer) the net present value (NPV) of future fuel savings per vehicle minus the incremental cost per vehicle of fuel economy technology. Following the lead of Ahmad and Greene (1993) we simplify by assuming that each vehicle design is essentially fixed except for its fuel intensity. If the initial level of fuel intensity is G^0_{mvi}, and the fractional change in fuel intensity is X_{mvi}, then the firm objective for each vehicle i in class v can be written as a linear expression for fuel savings minus a quadratic function for fuel economy technology cost:

$$\begin{aligned} NPV(X_{mvi}) &= K_v \cdot \left[G^0_{mvi} - G^0_{mvi}(1.0 + X_{mvi})\right] - \left[b_{mv} X_{mvi} + c_{mv} X^2_{mvi}\right] \\ &= -K_v \cdot G^0_{mvi} X_{mvi} - \left[b_{mv} X_{mvi} + c_{mv} X^2_{mvi}\right] \end{aligned} \tag{3}$$

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[14]Thus, a car achieving the 27.5 mpg standard is equivalently using 3.64 gallons per hectomile. A light truck achieving the 20.7 mpg standard is using 4.83 gallons per hectomile.

[15]In this equation and elsewhere we suppress the model index i when referring to the sum over all vehicle models i in class v for manufacturer m, with the understanding that $S_{mv} \equiv \sum_i S_{mvi}$. We also write $G_{jv}* = G_v*$, since all manufacturers face the same fuel intensity (performance) standards.

where parameter K_v is the estimated present value of fuel savings over the lifetime of a typical vehicle in class v for a unit change in fuel intensity (the units of K_v are ($/veh)/GPHM)).

The number of credits produced (number sold net of purchases) Z_{mv} by a manufacturer m is equal to the credit allowance minus the credit demand.[16] That is, the difference between the fuel intensity standard G_v* and the achieved average fuel intensity of its new vehicle fleet G_{mv}, times the total number of vehicles it produced, $S_{mv} = \sum_i S_{mvi}$ in class v. We write the achieved fuel intensity G_{mvi} as the original fuel intensity G^0_{mvi} times one plus the fractional change in intensity X_{mvi}. Let P_v be the price of a fuel use credit for vehicle type v denominated in units of dollars per vehicle-gallons per 100 miles ($/veh-GPHM). That is, P_v is the price per vehicle of relaxing the fuel economy constraint by 1 gallon per 100 miles of travel. The competitive manufacturers problem is:

$$\begin{aligned} &\underset{X_{mvi}, Z_{mv}}{Max} \quad \sum_i NPV(X_{mvi}) \cdot S_{mvi} \; + \; P_v Z_{mv} \\ &s.t.: \quad Z_{mv} \; = \; S_{mv} \cdot G_v^* \; - \; \sum_i S_{mvi} \cdot G^0_{mvi}(1 \; + \; X_{mvi}) \end{aligned} \qquad \textbf{(4)}$$

Under credit trading, each manufacturer m produces a set of vehicles indexed by i that are in regulated class v, adjusting their fuel intensities to maximize the net value of fuel use reductions *plus* the revenues from fuel use credits sold (or purchased) in credit market v. Note that each credit market v, that is each group of vehicle models, classes and manufacturers that may pool and exchange credits, will have its own credit price P_v.

To solve for the outcomes for all manufacturers, the set of problems for each firm as stated above must be supplemented by overall market constraints on credit balances. The scope and nature of the credit trading market can be represented by sign restrictions on credit production or various sums of credit production across vehicle classes or manufacturers, as shown in Table 1. Finally, because a positive

[16] We use the sign convention that when $Z_{mvi} > 0$ net credit production is positive.

market price for credits can only be sustained if the market constraint on credit balances is actually binding, the market solution, including the determination of credit prices, also requires complementary slackness conditions

Table 1: Summary of Trading Cases and Solution Conditions

Case	Description	Trade Among Veh Classes ?	Trade Among Firms ?	Credit Constraint	Complementary Slackness Cond.
1	No trading among firms or vehicle classes, with a separate standard G_v^* for each vehicle class (corresponds to the current class-based CAFE standard)	N	N	Z_{mvi} *u.i.s.* (*unrestricted in sign*). $Z_{mv} \equiv \sum_i Z_{mvi} \geq 0 \;\; \forall m,v$	$P_{mv} \cdot Z_{mv} = 0$; $P_{mv} \geq 0 \;\; \forall m,v$
2	Class Averaging: trading among vehicle classes within each firm, but not between firms. Corresponds to eliminating the vehicle class distinction from current CAFE standard.)	Y	N	Z_{mvi}, Z_{mv} *u.i.s.*, but $Z_m \equiv \sum_v Z_{mv} \geq 0 \quad \forall m$	$P_m \cdot Z_m = 0$; $P_m \geq 0 \quad \forall m$
3	Firm trading within classes (separate standard G^*_v for each vehicle class)	N	Y	Z_{mvi}, Z_{mv} *u.i.s.*, but $Z_v \equiv \sum_m Z_{mv} \geq 0 \quad \forall v$	$P_v \cdot Z_v = 0$; $P_v \geq 0 \quad \forall v$
4	Full (Firm & Class) Trading	Y	Y	$Z_{mvi}, Z_{mv}, Z_m, Z_v$ *u.i.s.*, but $Z \equiv \sum_{m,v} Z_{mv} \geq 0$	$P \cdot Z = 0$; $P \geq 0$

Consider first Case 3, credit trading in separate markets for each vehicle class v (other cases follow analogously). The first order conditions for this problem yield, for manufacturers behaving competitively in the credit market (i.e., for firms behaving as if $dP/dZ_m = 0$):

$$\frac{\partial NPV_{mv}}{\partial X_{mvi}} = P_v G^0_{mvi}, \qquad \forall \; m,v,i \tag{5}$$

The left hand side of **(5)** is interpreted as the marginal net present value per vehicle of a change in fuel use of a particular manufacturer's vehicle model. This must be equal to the price of a credit for fuel use weighted by the base fuel use for that model. We expect that at the optimum d*NPV*/d*X* will be positive: the CAFE constraint is binding and relaxing fuel intensity yields greater avoided technology costs than increased fuel costs. Vehicle production (S_{mvi}) drops out of the optimality condition because both the marginal value of fuel intensity and the marginal cost of credits are proportional to vehicle production.

For a competitive manufacturer, the credit price will equal the marginal cost of producing a credit. The credit price will be non-zero if the credit constraint (the aggregate fuel economy constraint for members of the credit market) is binding. Note that $-G^0_{mv}$ is the marginal change in credit supply per unit increase in fuel intensity, that is $\partial Z_{mv}/\partial X_{mvi} = -S_{mvi}G^0_{mvi}$. Thus

$$P_v = \left(\frac{\partial NPV_{mvi}}{\partial X_{mvi}}\right)\Big/G^0_{mvi} = -\left(\frac{\partial S_{mvi}NPV_{mvi}}{\partial X_{mvi}}\right)\Big/\left(\frac{\partial Z_{mvi}}{\partial X_{mvi}}\right) = -\left(\frac{\partial S_{mvi}NPV_{mvi}}{\partial Z_{mvi}}\right) \quad \forall m,v,i \qquad \textbf{(6)}$$

Stated another way, we see that at the optimum, each *competitive* manufacturer adjusts the fuel intensity of its vehicles *i* to balance the marginal cost of producing another credit with the credit price. If the aggregate fuel intensity constraint over the whole tradeable credit market is non-binding, the credit price will fall to zero. Manufacturers will then alter their fuel intensity until their marginal net benefit is zero.[17]

In summary, we can state that with a competitive market for fuel economy credits it is optimal for manufacturers to sell or buy credits as long as the market price is higher or lower than their own marginal cost of providing any given level of net fuel economy benefit. In competitive equilibrium, marginal net fuel economy benefits are equalized across all manufacturers.

[17] Some manufacturers may be expected to increase fuel intensity in this case.

2.7 Private Market Model with Market Power in Credits

Cournot-Nash Strategy for CAFE Credits

Models of oligopoly require specific assumptions on the behavior of the actors. Well known analytical solutions exist for the special case of the duopoly: the Cournot solution, in which the two suppliers act simultaneously by anticipating the other's reaction function, and the Stackelberg solution in which one supplier takes the price leadership in anticipating the other's reaction function. In the Stackelberg case, the oligopolist offers thereafter a maximum profit supply quantity. Based on market projections on the supply side, we anticipate that Japanese manufactures are potential Stackelberg actors. Technically, Japanese manufacturers would reduce their quantity of credits sold to the market to achieve maximum profits. Other oligopolistic approaches are n-actor cooperative and non-cooperative games. In all cases, solutions depend critically on behavioral characteristics that are difficult to determine.

In the models of imperfect competition and credit trading, it is typical for the market price of credits to be a function of the difference between the total allotment of credits, exogenously set by a regulator, and those used by the dominant firm.[18] With fuel use credits, however, the total number of credits is determined based on a performance standard set by the NHTSA and sales S_{jvi} and fuel economy X_{jvi} of vehicles chosen by vehicle manufacturers. The price of credits, will nonetheless, still be a function of the level of net credit sales Z_{kv} (sales less purchases) by the dominant firms k.

We partition the set of manufacturers M into a subset of oligopolists, M_o, and a subset of competitive ("fringe") firms, M_F. Following the approach of Westskog (1996), we let each Cournot oligopolist player $j \in M_O \subset M$, take as given the net supply Z_k of fuel economy credits by other Cournot players (for $k \neq j$) and recognize the competitive firms' price taking behavior.[19] The price-taking behavior of the competitive fringe implies that the market price of credits is a function $P(Z_O)$ of the total

[18]See for example, Hahn (1984), Innes et al. (1991) and Westskog (1996).

[19]This assumes that there are no negotiations between manufacturers, i.e., no cooperation.

net supply of credits by oligopolistic firms Z_O.

The profit-maximizing problem for a non-competitive firm j is then to determine the change X_{jvi} in fuel intensity for each of its vehicles, and the total supply of fuel economy credits Z_{jv} to maximize vehicle value plus credit sales revenue:

$$\begin{aligned}
&\underset{X_{jvi},Z_{jvi}}{Max} \sum_i [NPV(X_{jvi})\cdot S_{jvi} + P_v Z_{jvi}] \\
&st: \\
&P_v = P_v\left(\sum_{k\in M_O} Z_{kv}\right) = P_v(Z_{jv} + Z_{\sim jv}) \qquad \forall\ v,\ j\in M_O \\
&Z_{jvi} = S_{jvi}\cdot G_v^* - S_{jvi}\cdot G_{jvi}^0(1 + X_{jvi}) \\
&Z_{jv} \equiv \sum_i Z_{kvi} \\
&Z_{\sim jv} \equiv \sum_{k\in M_O, k\neq j} Z_{kv} = fixed\ if\ Cournot
\end{aligned} \tag{7}$$

Assuming vehicle production quantities S_{jvi} (and therefor shares) are fixed, but initial fuel intensity $G_{jvi}{}^{0}$ is varied by the fractional change X_{jvi}, the Lagrangian first order conditions yield the non-competitive analog to Eq. **(21)**:

$$MV\ Permit_{jv} = \frac{\partial NPV_{jv}/\partial X_{jvi}}{G_{jvi}^0} = \left[P_v + \frac{\partial P_v}{\partial Z_{jv}} Z_{jv}\right] \quad \forall\ j,v,i \tag{8}$$

The left hand side of **(8)** can be interpreted as the marginal net present *economic* value of an fuel intensity credit for a manufacturer *j*'s vehicle class *v*. That is, it is the marginal economic value of a fractional change in fuel intensity of all models of class *v* ($dNPV_j/dX_{jvi}$) divided by the marginal number of credits needed per unit-change in fuel intensity ($dZ_{jv}/dX_{jvi} = G_{jv}{}^{0}$). For a net credit seller, $Z_{jv} > 0$, this marginal value of a credit must be equal to the marginal revenue from selling an additional fuel use credit times the market share of that vehicle model and it's original fuel use (intensity). Since $[P_v + (\partial P_v/\partial Z_{jv})Z_{jv}] < P_v$ for credit sales from a firm with market power, this means that there is less incentive to decrease the fuel intensity of a manufacturer's fleet of vehicles (and thereby earn credit revenues) as compared to a competitive credit market. For a net credit buyers, $Z_{jv} < 0$, the opposite result obtains for price, $[P_v + (\partial P_v / \partial Z_{jv}) Z_{jv}] > P_v$. Here, vehicle manufacturers face higher prices of fuel use credits than under a competitive market and thereby purchase fewer fuel use credits. Thus, market power in the fuel use credit market causes both oligopolistic buyers and sellers to produce and consume fewer fuel intensity credits as compared to the competitive market situation.

This result is similar to those of Hahn (1984) and Westskog (1996). One important difference is that in their models, the total number of credits is set by a regulator. They note that, in principle, a regulator could ameliorate market power by assigning firms with market power the number of credits that the firms would want to hold after trading takes place. A variation of this solution is available in this market, regulators could assign different fuel intensity requirements on manufactures. This is because there is no set number of credits issued in this market; the credits are defined in terms of intensity, not an absolute number per period. The result is the regulatory structure which regulates efficiency, and leaves the number of vehicles sold by each manufacturer unregulated.

Implementation of the Cournot-Nash Solution

We implement the Cournot-Nash solution extending the approach of Westskog (1996). In her model there is a residual demand for credits from competitive fringe firms, $f \in M_F \subset M$, that take credit prices as given. With this distinction between the sets of fringe firms M_F and Cournot oligopoly firms M_O we have $M_F \cup M_O = M$, and total fringe demand for credits is:

$$Z_{Fv}(P_v) = \sum_{f \in M_F} Z_{fv}(P_v), \quad where \ P_v = P_v(Z_{Fv}) \tag{9}$$

From **(5)**, we know that a competitive fringe firm *f* will change fuel intensity until the marginal net cost of generating a credit is equal to the credit price. Taking the derivative of the net benefit function per-vehicle, we get the fringe's inverse demand curve for credits

$$P_v(Z_{fv}) = -\left[k_{1v} G_{fv}^0 + b_{fv} + 2c_{fv} X_{fvi}\right] / G_{fv}^0 \tag{10}$$

where, k_{1v} represents the effective discounted vehicle lifetime value of fuel use of vehicle class v. Parameters b and c represent a quadratically increasing cost of fuel technology as fuel intensity is reduced via adding more efficient vehicle technologies.[20] Solving for X_{fv} yields fringe firm f's optimal fuel intensity change (percentage increase) for vehicle i class v, as a function of credit price.

$$X_{fvi}(P_v) = -\frac{b_{fv} + [k_{1v} + P_v]G_{fv}^0}{2c_{fv}} \tag{11}$$

Then, using our expression for the net supply of credits **(3)** we can solve for each fringe firm f's demand for credits Z_{fv} for vehicles of class v in terms of the credit price set via the Cournot firms. Summing over the individual fringe firms and vehicle classes yields an aggregate demand for credits from the fringe.

$$Z_{Fv}(P_v) = \sum_{f \in M_F} Z_{fv}(P_v) = \sum_{f \in M_F}\left[G_v^* S_{fv} - \sum_i G_{fv}^0 \cdot S_{fvi}\left[1 - \frac{b_{fv} + [k + P_v]G_{fv}^0}{2c_{fv}}\right]\right] \tag{12}$$

The units for fuel intensity are gallons per hundred miles and the units for credits are vehicle-gallons per hundred miles (veh-GPHM). We can group terms and simplify to highlight that the total fringe supply is a linear function of price,

$$\begin{aligned} Z_{Fv}(P_v) &= Z_{FSv}^* - Z_{FD0v} + \Delta Z_{Fv}(0) + \alpha_v P_v \\ Z_{FSv}^* &\equiv \sum_{f \in M_F} G_v^* S_{fv} \\ Z_{FD0v} &\equiv \sum_{f \in M_F}\sum_i G_{fv}^0 \cdot S_{fvi} \\ \Delta Z_{Fv}(0) &\equiv \sum_{f \in M_F}\sum_i G_{fv}^0 \cdot S_{fvi}\left(\frac{b_{fv} + kG_{fv}^0}{2c_{fv}}\right) \\ \alpha_v &\equiv \sum_{f \in M_F}\sum_i G_{fv}^0 \cdot S_{fvi}\left(\frac{G_{fv}^0}{2c_{fv}}\right) \end{aligned} \tag{13}$$

The newly-defined terms in this equation correspond for vehicles of class v to the total credits allocation to the fringe Z_{FSv}^*, total initial credit demand (at base intensity) for the fringe Z_{FD0v}, fringe net

---

[20]We drop the subscripts on the b and c cost parameters for notational ease. In the simulations, b and c are manufacturer and vehicle class specific.

supply if credit price were zero $\Delta Z_{Fv}(0)$, and the rate of credit supply increase with price, α_v.

We invert the linear supply function to yield the price function for credits from the totality of fringe firms:

$$
\begin{aligned}
&P_v(Z_F) \;=\; \frac{\beta_v}{\alpha_v} + \frac{1}{\alpha_v} Z_{Fv}, \quad \textit{where,} \\
&\beta_v \;\equiv\; -\left(Z^*_{FSv} \;-\; Z_{FD0v} \;+\; \Delta Z_{Fv}(0)\right)
\end{aligned}
\tag{14}
$$

In the oligopoly-with-competitive fringe model, each oligopolistic firm *j* anticipates the effect of its production on total supply, and thereby on market price. Oligopolist firms know the credit supply response of the fringe, Z_F, and make a conjecture about the response of other oligopolistic firms, $Z_{o\sim j}$. Each oligopoly firm *j* recognizes the balance constraint for the market in credits (all Z's represent *net* supply from firms in the credit market):

$$
\begin{aligned}
&Z_{Fv} \;+\; Z_{jv} \;+\; Z_{O\sim j,v} \;=\; 0, \quad \forall \quad v, j \in M_O \\
&\text{where,} \qquad Z_{Fv} \;\equiv\; \sum_{f \in M_F} Z_{fv} \\
&\qquad\qquad Z_{O\sim j,v} \;\equiv\; \sum_{k \in M_O,\, k \neq j} Z_{kv}
\end{aligned}
\tag{15}
$$

From the balance equation the oligopolist *j* can infer how fringe supply must vary for an increase in his supply Z_j:

$$
\frac{dZ_{Fv}}{dZ_{jv}} \;=\; -1 \;-\; \frac{dZ_{o\sim j,v}}{dZ_{jv}}
\tag{16}
$$

A critical assumption of any oligopoly model is the "conjectural variation" cv_j or assumed response of other oligopoly firms to a change in supply from firm *j*, denoted $cv_j \;\equiv\; \partial Z_{o\sim j}/\partial Z_j$. In the Cournot oligopoly model the hypothesized conjectural variation is zero, hence $dZ_{Fv}/dZ_{jv} = -1$, and

$$
\frac{dP_v}{dZ_{jv}} \;=\; \frac{dP_v}{dZ_{Fv}} \frac{dZ_{Fv}}{dZ_{jv}} \;=\; -\frac{dP_v}{dZ_{Fv}}
\tag{17}
$$

Using this Cournot anticipated price response, and the fringe inverse supply curve **(12)** in the oligopolist's first order conditions for profit maximization, **(8)**, we get the following necessary condition for each oligopolist.

$$-\frac{[k_{1v}G_{jv}^0 + b_{jv} + 2c_{jv}X_{jv}]}{G_{jv}^0} + \left[\frac{1}{\sum_{f\in M_F}\sum_i\left(\frac{(G_{fv}^0)^2 S_{fvi}}{2c_{fv}}\right)}Z_{jv}\right] = P_v, \qquad \forall\ v, j\in O \tag{18}$$

This establishes the optimal behavior of Cournot oligopolists with respect to price.

Thus, a Nash solution to the Cournot oligopoly problem is to simultaneously satisfy the equations in **(18)** and **(10)** by equating all of the left-hand sides to one-another, and the credit balance equation $\sum_{m\in M} Z_{mv} \geq 0.$ Additionally, we impose the complementary slackness conditions shown in Table 1 to address the cases in which the credit price collapses to zero.

3 Model Parameterization

3.1 Parameterization for Fuel Savings

We need to estimate the parameter K_v, that represents the consumer's present discounted value of fuel economy of vehicle class *v*. Given that vehicles have a relatively short lifespan for a major capital expenditure, we assume that consumers treat fuel economy technology as a depreciating asset. This implies that the consumer will demand a higher rate of return for an investment in fuel economy than for an investment in a non-depreciating asset. The rate of return consumers will demand for fuel economy improvements will be primarily determined by the expected life of the vehicle, L_v, and the rate of decline in use of the vehicle.[21] Although higher rates of return on fuel economy investments could be argued for,

[21]Data from Oak Ridge National Laboratory show that the median lifetime for a 1990 vintage car or truck is 16.9 and 15.5 years respectively (Davis and Diegel, Tables 3.6 and 3.7). National survey data indicate that new private automobiles and trucks travel 15,000 and 17,500 miles, respectively, in their first year of operation (Davis and Diegel, Tables 3.6 and 3.7). However, these same data shows that vehicle use (miles driven) declines with vehicle age, which implies declining annual fuel savings. We take as a reasonable approximation in the rate of decline in use for cars and trucks to be 4.0 and 3.0 percent respectively (USDOT, 2004, Davis and Diegel).

12%/year will be used as a base case assumption for this analysis.

Clearly, consumers do not know what future fuel prices will be. We model consumers as having static expectations over fuel prices. That is, consumers will assume that the future price of fuel will be the same as the current price at the of vehicle purchase. We use the Energy Information Administration's 2012 reference case 2012 forecast price of $1.51 and "high B" forecast of 1.84 cents per gallon (EIA, 2005, Table 12). The price of fuel *P* is unaffected by choices about vehicle fuel economy, and average vehicle economy for the fleet of new vehicles.

We make the additional assumption that the utilization of each vehicle (vehicle miles traveled per year) *M* is fixed for each vehicle class, regardless of choice of fuel economy F. If vehicle owners drive more with a higher fuel efficiency vehicle then we are underestimating the value of fuel economy purchased.

Given these estimates we are now able to estimate K_v the consumer's present discounted value of fuel economy of vehicle class *v* as the lifetime discounted miles driven, M_v, times the current fuel price in year y, P_y, applying the declining use rate γ_v and the discount rate ρ_v. Note that we divide the number of miles through by κ=0.85 to discount test-value mpg numbers to reflect real-world performance.[22]

$$K_v = \frac{PM_v}{\kappa}\left(\frac{1.0}{\gamma_v + \rho_v}\right)\left(1.0 - e^{-(\gamma_v + \rho_v)L_v}\right). \tag{19}$$

Only the monetary costs and benefits of fuel economy will be considered since fuel economy technologies are assumed hedonically neutral. That is, except for their impacts on fuel economy and vehicle price, they do not enter into a consumer's purchase decision or affect a consumer's satisfaction with a vehicle. Thus, our base case cost curves do not include diesel and hybrid technology. While this may understate the fuel economy technology available, the NAS technologies are nearly invisible to the

[22] Consumers typically achieve lower fuel economy in actual on-road driving than the EPA dynamometer test MPG numbers (e.g., Hellman and Murrell, 1984). Although there is evidence that the shortfall for trucks may be larger than that for passenger cars (Mintz et al., 1993), the average shortfall of 15% implied by EPA official correction factors is used here for both vehicle types.

consumer. This is not necessarily true for diesels and hybrids. These technologies may penetrate the market in different ways in terms of consumer tradeoffs. Diesel and hybrid technologies can be added to the NAS list, but they will be disruptive, superior, to the other technologies on this list. Changes in the hedonic value due to changes in vehicle attributes are not only difficult to predict, but difficult to value, as well. [23]

3.2 Fuel Intensity Cost Curves

We use data for MY 2003 vehicles sold in the United States, obtained from the National Highway Traffic Safety Administration (NHTSA) Manufacturer's Fuel Economy Reports. These give us vehicle manufacturers sales and fuel economy by vehicle class (8 cars and 7 truck) and country of origin (foreign or domestic). Not all manufacturers have product offerings in all vehicle type categories. Moreover, examination of the weight and horsepower, and fuel economy data also confirms that manufacturers' product offerings differ somewhat even within vehicle size/class categories. For example, the average fuel economy of a compact car from BMW is lower than that of Ford.

The National Research Council's presents low and high retail equivalent price estimates for a low and high range of incremental fuel efficiency gains by individual technologies for 4 car and 6 truck classes (NRC, Tables 3.1 -3.4) .[24] In particular, we use National Research Council's "emerging" (path 3) technologies . Except for camless valve actuation and variable compression ratio technologies, the rest of the technologies are either implemented on some vehicles now or are capable of being implemented in the time frame of the National Research Council's analysis. This is conservative assumption because it does

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[23] Beyond these practical reasons, we also do not include hybrid and diesel vehicles since they are still expected to make up only a small amount of the market in the time frame of our analysis. The EIA's Annual Energy Outlook 2005 reference case projects hybrid car and diesel new car sales to be 5.8% and 0.3% by 2016 (EIA AEO, p. 29). The EIA notes that regulations pending implementation in California would regulate the greenhouse gas emissions of light-duty vehicles in California. If this legislation is also adopted by other states that have adopted California vehicle emission standards (New York, Maine, Massachusetts, and Vermont), this could lead to a 11.0% and 0.9% of total new car sales to be hybrid and diesel by 2016 (EIA AEO, p. 29). These regulations are currently being challenged in federal court.

[24]For cars and trucks these include: subcompact, compact, midsized, large, (trucks) SUV-Small, SUV-Mid, SUV-Large, Minivan, Pickup-Small, Pickup-Large.

not include diesel or hybrid technology.

We use the National Research Council's high and low retail costs with low and high efficiency gains to generate low, average and high retail costs of fuel efficiency improvements that encompass the full range of cost and performance uncertainty. Before mathematical functions are fitted to the data, the technologies are ranked by a cost-effectiveness index, equal to the percent improvement in fuel economy divided by the price increase. This procedure ensures that technologies are implemented in order of increasing marginal cost, in accordance with economic theory. Engineering knowledge and judgment is also employed to ensure that combinations of technologies do not violate technological feasibility. The technology cost curves we develop, therefore, represent an aggregate description of the industry's ability to supply fuel economy, rather than a technical plan for improving the fuel economy of a particular vehicle. This generates low, average and high cost curves for 4 car and 6 truck class of vehicles.

A recent review of the technology cost literature indicated that two-parameter quadratic curves fit data from all studies reasonably well (Greene and DeCicco, 2000). The two-parameter quadratic cost function is shown in **(20)**.

$$P(X) = bX + cX^2 \qquad \textbf{(20)}$$

P(x) is the retail price (cost) increase to the percentage decreases in gallons per 100 miles over a base level, G^o , and b and c are parameters to be estimated. By construction, the curves pass through the origin (0% improvement has $0 cost). The parameter estimates are intended to be curve fits and not statistical estimations. The important point is that the fitted curves accurately reflect the rate of increase in retail price for a percent decrease in fuel intensity for the full range of fuel economy improvements being considered. This generates a high, average and low cost curve for each the National Research Council's 10 vehicle types. The two-parameter quadratic functions fit the data very well, with adjusted R-squared values exceeding 0.98 in all instances.[25]

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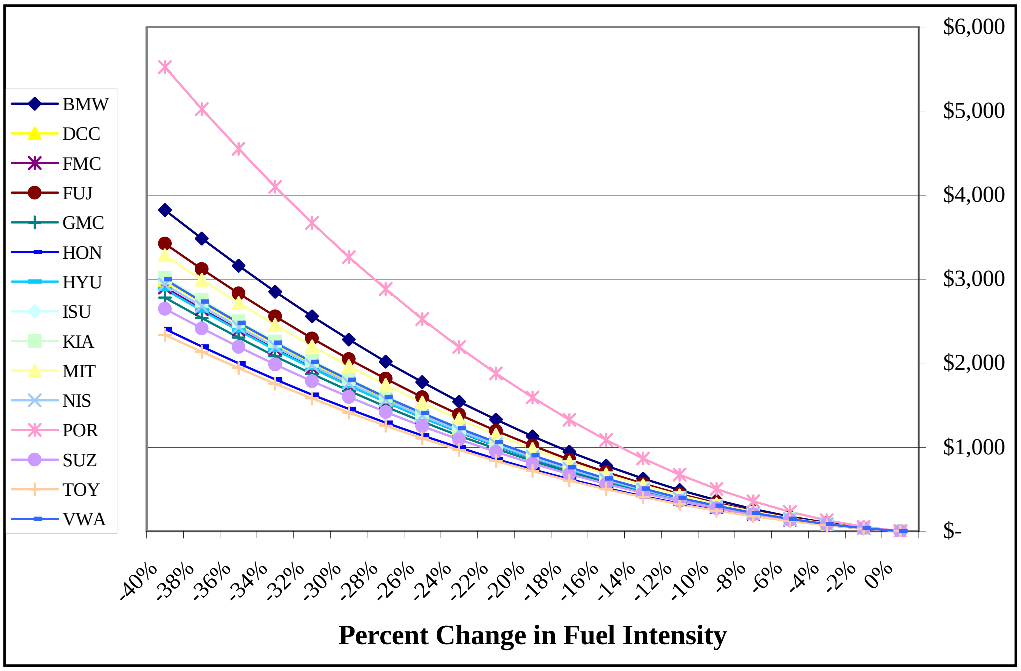
[25]For a few of the low cost, high decrease in fuel intensity cases we dropped a outlier data points at the high end to improve the fit of the curves for low decreases in fuel intensity range.

We then generate manufacturer-specific cost curves by weighting both of the estimated coefficients for the 10 size classes and vehicle types by the manufacture specific sales-weighed average of vehicles and fuel intensities. For example, to generate a particular vehicle manufacturer's cost curve for cars, we combine the sales-weight average of the parameters for the 4 size classes produced by that manufacturer and weighted by fuel intensity of that manufacturer. We show of estimates for the average cost and performance case in Figure 1.

Figure 1: Fuel Intensity Costs by Manufacturer (Average of Costs for Cars and Trucks)



4 Results

4.1 Percentage Cost Savings

In order to explore the potential cost savings from allowing more regulatory flexibility by credit trading we examine 4 possible credit trading scenarios. These scenarios reflect increasing amounts of flexibility, starting from the base case that does not allow any credit trading by manufacturers consistent with current CAFE regulations (see Table 2).

Table 2: Credit Trading Scenarios

Scenario Name	Credit Trading Among Firms	Credit Trading Among Vehicle Classes	Scenario Description
Base (No Trading)	No	No	Firms must independently meet separate standards for cars and trucks
Class Averaging	No	Yes	Firms trade credits across vehicle classes (but not among firms)
Class Trading	Yes	No	Credits trade amongst firms but in separate car and truck markets
Firm & Class Trading	Yes	Yes	Firms can trade credits in a single a single market

In addition, for those scenarios that allow credit trading among manufactures, we allow varying degrees of competition, from an assumption of perfect competition to the case where we assume that our 5 largest competitors (General Motors, Ford, DaimlerChrysler, Toyota and Honda) each act as independent oligopolists. To test the sensitivity of our parameter assumptions we also examine each of these cases assuming low and high valuation of fuel economy by consumers, base and high projects of gasoline prices, and low, medium and high costs and effectiveness of the fuel economy technology. Given the large number of permutations of cases, we focus on deviations from our base case: no credit trading among vehicle type or manufacturers, low valuation of fuel economy cost savings by consumers, average costs of fuel economy technology and base projections of fuel prices. This scenario is closest to representing the current CAFE regulations with conservative assumptions concerning the valuation of fuel economy and average cost assumptions. One minor point in which we differ from the current regulatory outcome, is that we insure that each manufacturer does comply with the CAFE regulations rather than fall short and pay the fines as noted earlier.[26]

[26]Historically, only BMW, Porsche and the manufacturers of a few other specialty high-performance cars actually have paid fines rather than meet the standard (NHTSA, 2005)

Table 3: Percentage Cost Savings due to Trading
(Base Case with Perfect Competition - High Fuel Economy Target*)

Scenario Name	Base Case	Low Cost of Fuel Economy Technology	High Cost of Fuel Economy Technology	High Future Gasoline Prices
No-trading Cost ($/vehicle)	*+$44*	*-$374*	*+$806*	*-$44*
Class Averaging	90%	1%	14%	86%
Class Trading	56%	1%	8%	50%
Firm & Class Trading	128%	1%	19%	119%

*Base case: no credit trading among vehicle type or manufacturers, low valuation of fuel economy cost savings by consumers, average costs of fuel economy technology and base projections of fuel prices, all firms joint net benefits and a 40% increase in fuel economy by 2015. Savings are percentage cost reductions relative to the cost of increasing fuel economy under the no-trading baseline (current policy).

Shown in data column 1 of Tables 4 and 3 are the percentage costs savings to all manufacturers taken together from allowing trading of fuel economy credits under our base case assumptions and assuming perfect competition. The difference between the two tables is the level of increased fuel economy required; Table 4 assumes and increase of 30% and Table 3 assumes an increase of 40% by 2015. What stands out, as expected, is that the highest level of regulatory flexibility, firm and vehicle class trading, yields the greatest savings. Given the construction of our cost curves and the market shares of the vehicle manufacturers, we find that “class averaging” (allowing vehicle manufacturers to trade fuel economy credits across their vehicle classes) provides the next greatest level of savings. This is followed by class trading among manufactures where manufacturers can sell or buy a car and truck credits with other manufacturers in separate car and truck markets.

That a significant portion of the total savings available is from class averaging within firms is of particular importance in term of possible non-competitive behavior. This portion of savings will not be affected by the possible oligopolistic or oligopsonistic withholding of credit trades from the market in order to drive credit prices up or down. Note that adding the percentage saving between class averaging and class trading yields a greater level of savings than complete flexibility that allows both of these trades to occur simultaneously. This shows that the regulatory flexibility of class averaging and class trading are, to some extent, substitutes. However, the magnitude of the substitution effect does not appear great.

As shown in data columns 2-4, the magnitude of the percentage savings depends substantially on the particular scenario under examination, but the same pattern of savings across the trading systems remains unchanged. What is clear is that if our base scenario assumptions are correct, the cost savings from fuel economy credit trading are quite substantial, possibly in excess of 100% of the net costs of increasing fuel economy by 30% or 40% under the current regulatory regime.

Table 4: Percentage Cost Savings due to Trading
(Base Case with Perfect Competition - Low Fuel Economy Target*)

Scenario Name	Base Case	Low Cost of Fuel Economy Technology	High Cost of Fuel Economy Technology	High Future Gasoline Prices
No-trading Cost ($/vehicle)	*-$21*	*-$377*	*+$495*	*-$94*
Class Averaging	127%	0%	15%	25%
Class Trading	99%	0%	13%	18%
Firm & Class Trading	187%	0%	23%	36%

*Base case: no credit trading among vehicle type or manufacturers, low valuation of fuel economy cost savings by consumers, average costs of fuel economy technology and base projections of fuel prices, all firms joint net benefits and a 30% increase in fuel economy by 2015. Savings are percentage cost reductions relative to the cost of increasing fuel economy under the no-trading baseline (current policy).

Looking at the cost savings under alternative assumptions, a few points stand out. First, there is a very large range in the retail cost and the technological effectiveness presented in the NRC's data. As a result the low and high cost cases present very different views. In the low cost/high technological effectiveness case (data column 2), there are effectively no cost savings from trading because the imposed higher CAFE standards are essentially not binding on the industry as a whole. Similarly, the percentage gains from trading are significantly less under the high fuel economy cost/less effective technology case (data column 3) than the base case. This is because the manufacturers as a group have less ability to find savings through averaging and trading. They all must significantly increase the use of fuel economy technology. Thus, the added flexibility is still valuable in absolute terms, but the savings, as a percentage of overall costs from the no-trading baseline, are much reduced. The magnitude of the savings, not surprisingly, is therefore highly dependent upon the accuracy of the NRC's estimates of the costs and effectiveness of the fuel economy technology. Similarly, looking at the final column we see that higher future gasoline prices increase the value of additional regulatory flexibility (trading) by enhancing the value to consumers of the additional fuel economy technology.

As discussed earlier, given the large proportion of vehicles produced by the 5 largest manufacturers, the effect of market power in the price and availability of fuel economy credits needs to be examined explicitly. In Tables 6 and 5 we show the percentage cost savings from 3 different cases of imperfect competition relative to two different baselines, given the base case assumptions used earlier. The first column of each table repeats the savings shown above assuming perfect competition in the credit markets. The 3 non-competitive cases make different assumptions about which manufacturers act as oligopolists: all 5 major manufacturers, only Honda and Toyota, or only (what was formerly called) the big three US firms (Ford, General Motors, DaimlerChrysler). As before, the cost savings shown are for all manufacturers jointly. Now especially, since we examine the impact of market power, the gains to individual manufacturers from credit trading will vary. Individualized impacts are examined later.

Table 5: Percent Cost Savings Due To Trading

(Comparison Across Various Non-Competitive Cases - High Fuel Economy Target)

	Perfect Comp	All 5 Majors		Honda & Toyota		US Big 3	
Scenario Name	Relative to Base	Relative to Base	Relative to PC Case	Relative to Base	Relative to PC Case	Relative to Base	Relative to PC Case

Class Averaging	90%	90%	0%	90%	0%	90%	0%
Class Trading	56%	47%	-21%	55%	-2%	55%	-3%
Firm & Class Trading	128%	124%	-13%	128%	-1%	127%	-1%

For each case of imperfect competition, the first column indicates the cost savings relative to its *own baseline of no averaging or trading*, while the second column shows the cost savings *relative to the equivalent trading scenario under perfect competition scenario*. For example in Table 6 we see that when the big five each act as independent Cournot oligopolists, the savings from being able to average and trade is 69% compared to the no trading baseline. This is a reduction in savings of 15% from averaging and trading when the credit market is perfectly competitive.[27] Note that the savings due to class averaging (trading among classes within each firm) are not affected by non-competitive behavior; this is shown in the table by the zeros relative to the perfectly competitive case.

Since this scenario posits oligopoly sellers and oligopoly buyers with a competitive fringe, determining who is able to extract the most gains is a numerical question. As is seen in Tables 6 and 7, when only Toyota and Honda or only the US big three act as oligopolists, the losses in efficiency are quite small. As we see below, gains and losses from imperfect competition are larger for individual companies compared to the market as a whole.

Table 6: Percent Cost Savings Due to Trading

(Comparison Across Various Non-Competitive Cases - Low Fuel Economy Target)

	Perfect Comp	All 5 Majors		Honda & Toyota		US Big 3	
Scenario Name	Relative to Base	Relative to Base	Relative to PC Case	Relative to Base	Relative to PC Case	Relative to Base	Relative to PC Case

[27]Since the baselines are different in the two columns, the percentage changes should not add across columns.

Class Averaging	127%	127%	0%	127%	0%	127%	0%
Class Trading	99%	69%	-15%	97%	-1%	93%	-3%
Firm & Class Trading	187%	174%	-5%	187%	-0%	186%	-1%

4.2 Firm Compliance Costs

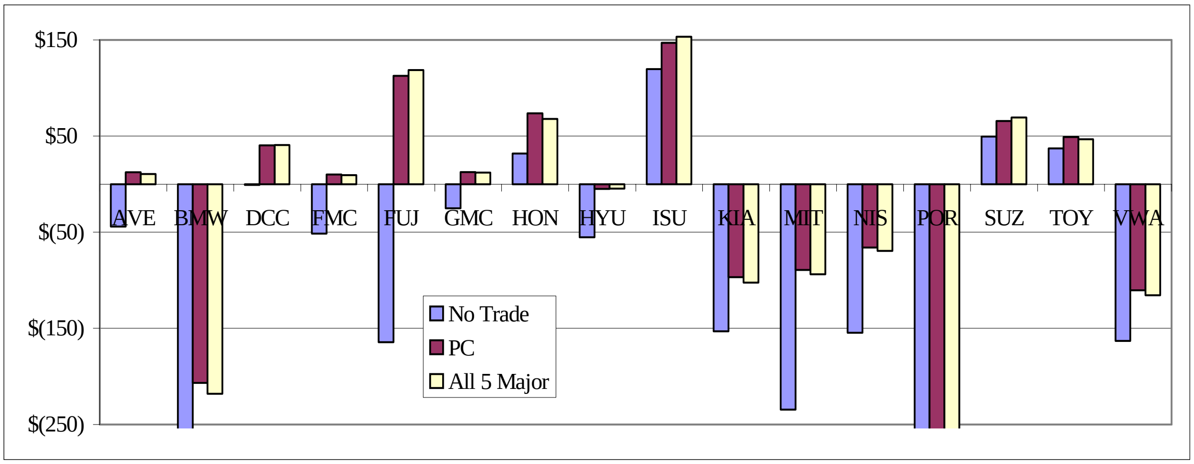


Figure 2: Net Present Value per Vehicle including Net Credit Sales (Base Case)

What matters from an individual firm's perspective is the net cost of compliance. Apart from technology costs and consumers perceptions of the value of future cost savings, these cost are determined from the degree of regulatory flexibility available and from the possible effects of non-competitive behavior in the market place. In Figure 2 we show the net revenues (positive and negative depending on manufacturer) of credit sales to the net technology and fuel costs. This figure captures our estimate of the average net total cost of compliance to a 40% increase in CAFE standards given the base case assumptions detailed above. For all manufacturers we see that allowing vehicle manufacturers to average and trade fuel economy credits lower the cost of compliance. As is seen in this figure, the magnitude of savings can be quite substantial for some manufacturers, less so for others. Importantly, for all manufacturers, both net sellers and net buyers, oligopolistic behavior, as we have modeled it, by the big five manufacturers does not substantially diminish the savings from being able to trade fuel economy credits.

Beside lowering the potential gains from credit trading, market power also affects the price of credits. Using the base case assumptions, with credit trading across classes and manufacturers yields the following credit prices for different scenarios of imperfect competition. These include assuming that all firms act perfectly competitively (PC), and the following groups act as Cournot oligopolists: Honda and Toyota (H&T), Ford, GM DaimlerChrysler (US 3), and Honda, Toyota, Ford, GM and DaimlerChrysler (All 5).

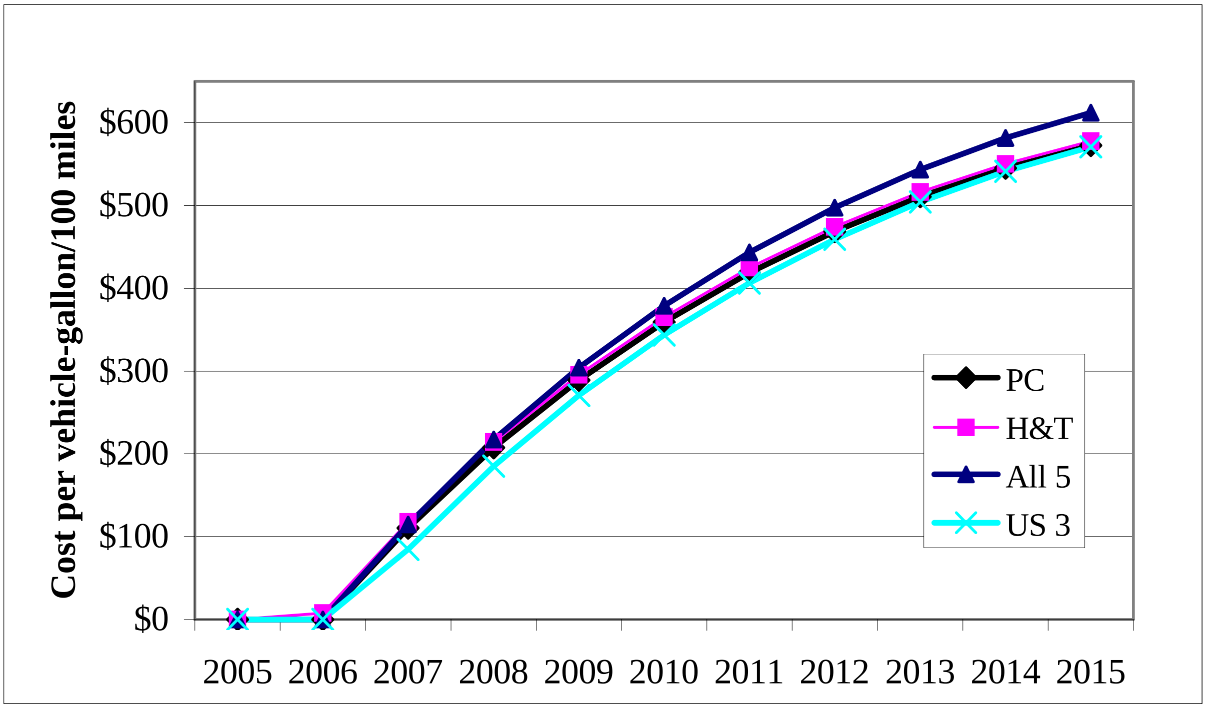


Figure 3: Credit Prices for Different Levels of Imperfect Competition

As is seen, the credit price is zero for the first year under all scenarios reflecting the gradually phased in stricter standards. As credit prices rise reflecting tighter standards, the divergence between the perfectly competitive price (PC) and the oligopoly prices grows. As expected, the credit price is slightly below competitive level when the net buyers ("US 3") act non-competitively, and slightly above the non-competitive price when the net sellers ("H&T") act non-competitively. The largest divergence occurs when the big five sellers each act as independent oligopolists (All 5).

Besides the market price of credits, however, is the behavior of firms, and therefore application of fuel economy technology, based on their marginal value of an additional credit. Fringe firms equate their marginal value of a credit to the price of credits. Cournot firms, however, following **(18)**, regardless of whether they are net buyer or sellers, anticipate the effect of their own purchases and sales on the market price. This divergence between the market price of credits and the marginal value to Cournot firms is the source of inefficiency that reduces the gains from trade available in a perfectly competitive market.

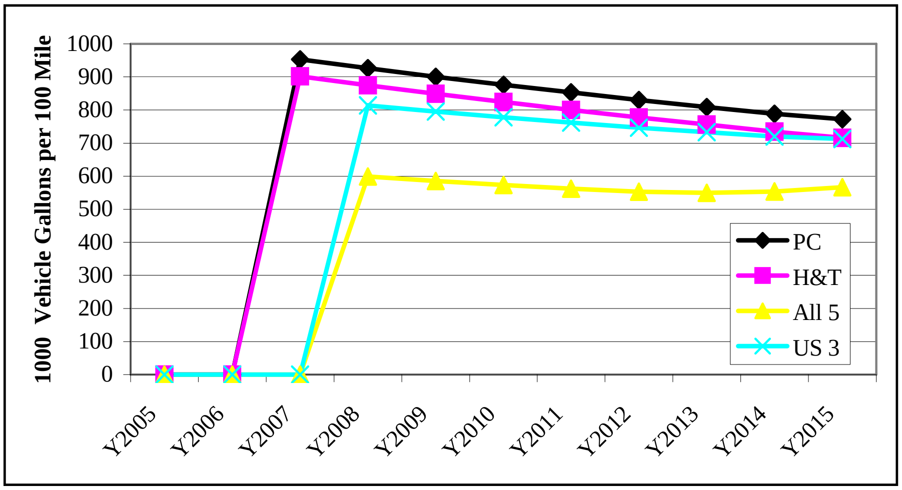


Figure 4: Credit Volume by Levels of Competition, Base Case

While the market price for credits could be higher or lower under oligopoly-verses-oligopsony trading than under perfect competition, depending on the relative market power of buyers and sellers, theory tells us the net effect of non-competitive supplier and demander behavior is always to reduce trade volumes. This is because both non-competitive buyers and sellers reduce their market transactions to limit their anticipated adverse effects on the market price of credits. This phenomenon is seen in Figure 4. Here in the most extensive case of market power we explore, all 5 major manufacturers behaving non-competitively, the credit volume drops about 35% compared to the perfectly competitive benchmark. Note credit volumes are zero for the first year or two reflecting a zero price for credits given that the standards are not initially collectively binding.

5. Final Comments

Depending upon the case, the net cost of tightened fuel economy standards to the industry as whole may be quite large or small. This uncertainty reflects the large range of possible costs of fuel economy technology, uncertain future gasoline prices, and ambiguity regarding how consumers value future fuel economy savings. The results in this paper show how the net costs also depend on the level of future fuel economy standards, the flexibility of the standards, and the degree to which a tradable credit market is affected by non-competitive behavior. Resolving uncertainty over the engineering costs of increasing fuel economy at the firm and industry level and improving our understanding of consumers' valuation of fuel economy is clearly needed.

For cost scenarios that impose significant costs on individual vehicle manufacturers, we find the savings from averaging and trading credits to be quite substantial. The greatest proportional savings exceeded 100% in some cases, reflecting the fact that, to the industry as a whole, average costs per vehicle went from a net negative to a net positive value. The scenarios that show large percentage gains from trade generally reflected the middle range of net costs - where there were substantial imposed costs on some, but not all, manufacturers from increased fuel economy standards. In cases where the net costs were lower (low cost of fuel economy technology or high consumer valuation of fuel economy improvements), the gains from trading were small or non-existent reflecting the largely non-binding nature of increased fuel economy targets. At the other extreme, when fuel economy improvements were expensive, the percentage gains from being able to average and trade credits were considerably smaller (while the absolute gains in dollars-per-vehicle were greater). For many of the scenarios, the ability of each manufacturer to average credits between its car and truck classes provides greater savings than the ability to only trade credits between manufacturers in separate vehicle class markets. As expected, the greatest savings comes from the greatest flexibility, when manufacturers are able to average and trade fuel economy credits.

Given the high concentration of vehicle sales by the five largest firms, we explicitly examined the potential impact of market power in the credit markets. We modeled the largest firms as Cournot oligopolists facing a competitive fringe. The theoretical effect of imperfect competition on fuel economy credit price (compared to a perfect competition benchmark) is ambiguous since firms with market power are both sellers and buyers. Our numerical simulations show that there is a small increase in the price of credits when all five of the largest firms act as oligopolists, and seek a Cournot-Nash equilibrium. However, both sellers and buyers of credits have an incentive to reduce their net credit transactions in order to influence to the credit price. We find that the volume of credit sales can be up to 35% less compared to the perfectly competitive benchmark.

As expected, the existence of market power did lower the potential cost savings to the industry as a whole. However, the magnitude of the potential losses in efficiency from the market power were not large, less than 25% of the potential savings from trade in all cases when considering the industry as a whole. Since some firms are net sellers and some net buyers, individual firms experienced greater gains or losses from trading when taking market power into consideration than did the industry as a whole. Importantly, every firm was still better off from credit trading with imperfect competition compared with our no trading baseline. Imperfect competition in credits does not appear to eliminate all the gains from trading at the firm level and has relatively modest impacts on the industry as a whole.

References

Ahmad, Sanjana and David Greene, "A Preliminary Economic Analysis of Tradable Credits for Fuel Economy Standards," Session 659, Annual Meeting, Transportation Research Board, National Research Council, January 15, 2003

Crandall, Robert W. and Graham, John D. 1989. "The Effect of Fuel Economy Standards on Automobile Safety," Journal of Law and Economics 32(1): 97-118.

Davis, Stacy C., 2002. *Transportation Energy Data Book*: Edition 22, ORNL-6967, Oak Ridge National Laboratory, Oak Ridge, Tennessee, June.

Davis, Stacy C. and Susan W. Diegel 2004. *Transportation Energy Data Book*, Edition 24, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Ellerman, A. D. and A. Decaux (1998), Analysis of Post-Kyoto CO2 Emissions Trading

Energy Information Administration, United States Department of Energy, *Annual Energy Outlook 2005 with Projections to 2025*, http://www.eia.doe.gov/oiaf/aeo/pdf/aeotab_12.pdf, and http://www.eia.doe.gov/oiaf/aeo/aeoref_tab.html.

Fershtman, Chaim and Aart de Zeeuw (1995), "Tradeable Emission Permits in Oligopoly," Tilburg Center for Economic Research, working paper 9630.

Fullerton, D. and G. E. Metcalf (2002). "Cap and Trade Policies in the Presence of Monopoly and Distortionary Taxation." Resource and Energy Economics 24(4): 327-347.

Godby, R. (2002), "Market Power in Laboratory Emission Permit Markets," Environmental and Resource Economics, 23:279-318.

Godek, Paul E., 1997. "The Regulation of Fuel efficiency and the Demand for Light-Trucks," Journal of Law and Economics, XL, October.

Greene, D.L. and J. DeCicco, 2000. "Engineering-Economic Analysis of Automotive Fuel Economy Potential in the United States," *Annual Review of Energy and the Environment*, 25:477-537.

Greene, David L. 1990. "CAFE or Price? An Analysis of the Effects of the Federal Fuel efficiency Regulations and Gasoline Price on New Car MPG, 1978-89," *The Energy Journal* 11(3):37-58.

Greene, David L. 1998. “Why CAFE Worked,” *Energy Policy* 26(8):595-613.

Greene, David L., 1991. “Short-Run Pricing Strategies to Increase Corporate Average Fuel Efficiency,” *Economic Inquiry* XXIX(January):101-114.

Hagem, C. and H. Westskog (1998). “The Design of a Dynamic Tradeable Quota System under Market Imperfections.” *Journal of Environmental Economics and Management* 36(1): 89-107.

Hahn, R. W. (1984), “Market Power in Transferable Property Rights,” *Quarterly Journal of Economics*, 99, 753-765.

Hellman, K.H. and J.D. Murrell. 1984. “Development of Adjustment Factors for the EPA City and Highway MPG Values,” SAE Technical Paper Series #840496, Society of Automotive Engineers, Warrendale, Pennsylvania.

Innes, Robert, Catherine Kling, and Jonathan Rubin, “Emission Permits Under Monopoly,” *Natural Resource Modeling*, v. 5(3) (Summer 1991), pp. 321-343.

Kleit, Andrew N. 1990. “The Effect of Annual Changes in Automobile Fuel efficiency,” *Journal of Regulatory Economics*, 2:151-172.

Kwoka, John Jr. 1983. “The Limits of Market-Oriented Regulatory Techniques: The Case of Automotive Fuel Efficiency,” *The Quarterly Journal of Economics*, November 1983.

Leiby, Paul and Jonathan Rubin, 2001. “Bankable Permits for the Control of Stock and Flow Pollutants: Optimal Intertemporal Greenhouse Gas Emission Trading,” *Environmental and Resource Economics*, 19: 229-256.

Malueg, D. A. (1990), “Welfare Consequences of Emission Credit Trading Programs,” *Journal of Environmental Economics and Management* 18, 66-77.

Mintz, M., A. D. Vyas and L.A. Conley. 1993. “Differences Between EPA-Test and In-Use Fuel Economy: Are the Correction Factors Correct?,” *Transportation Research Record* 1416, pp. 124-130, Transportation Research Board, National Research Council, National Academy Press, Washington, DC.

Misiolek, W. S. and H. W. Elder (1989), “Exclusionary Manipulation of Markets for Pollution Rights,” *Journal of Environmental Economics and Management*, 16 (2), 156-166.

National Commission on Energy Policy, 2004. “Ending the Energy Stalemate: A Bipartisan Strategy to Meet America’s Energy Challenges,” http://www.energycommission.org/.

National Research Council, 2002. “Effectiveness and Impact of Corporate Average Fuel Economy (CAFE) Standards,” National Academy Press, Washington, D.C.

National Highway Traffic Safety Administration, U.S. Department of Transportation, 2004, “Summary of Fuel Economy Performance,” http://www.nhtsa.dot.gov/Cars/rules/CAFE/CAFEData.htm

National Highway Traffic Safety Administration, U.S. Department of Transportation, 2005, "Summary of CAFE Fines Collected," http://www.nhtsa.dot.gov/Cars/rules/CAFE/fines-colleted-summary.pdf. (sic).

National Highway Traffic Safety Administration, U.S. Department of Transportation, 2006. “Average Fuel Economy Standards for Light Trucks Model Years 2008-2011,” 49 CFR Parts 523, 533 and 537.

Nivola, Pietro S. and Robert W. Crandall, 1995. *The Extra Mile: Rethinking Energy Policy for Automotive Transportation*, The Brookings Institution, Washington, D.C.

Parry, Ian W.H., Carolyn Fischer and Winston Harrington, 2004. “Should Corporate Average Fuel Economy (CAFE) Standards Be Tightened?, Resources for the Future Discussion Paper 04–53.

Pew Center on Global Climate Change, *Agenda for Climate Action*, February 2006

Rubin, Jonathan and Catherine Kling, “An Emission Saved is an Emission Earned: An Empirical Study of Emission Banking for Light-Duty Vehicle Manufacturers,” *Journal of Environmental Economics and Management*, 25(3) (November, 1993), 257-274.

Sartzetakis, E. S. (1997a), “Tradeable Emission Permits Regulations in the Presence of Imperfectly Competitive Product Markets: Welfare Implications,” *Environmental and Resource Economics*, 9 (1), 65-81.

Thorpe, Steven G. 1997. “Fuel efficiency Standards, New Vehicle Sales, and Average Fuel Efficiency,” *Journal of Regulatory Economics* 11(3):311-326.

Tirole, Jean (1990), *The Theory of Industrial Organization*, The MIT Press, Cambridge, Massachusetts, 4th edition.

Train, K. (1986) *Qualitative Choice Analysis*, MIT Press ,Cambridge, MA.

Transport for London, 2004a. “Congestion Charging: Update on Scheme Impacts and Operations,” February.

Transport for London, 2004b, “ Transport Annex E: Report to the Mayor, July 2004 Environmental Assessment. ”

US Department of Energy (1995) “Effects of Feebates on Vehicle Fuel Economy, Carbon Dioxide Emissions, and Consumer Surplus,” DOE/PO-0031, Office of Policy, Washington, D.C., February.

US Environmental Protection Agency, Office of Atmospheric Programs, 2005, Inventory of U.S. Greenhouse Gas

Emissions and Sinks: 1990-2003, EPA 430-R-05-003.

Weiss, M.A., J.B. Heywood, E.M. Drake, A. Schafer and F.F. Au Yeung, 2000. "On the Road in 2020," Energy Laboratory Report #MIT EL 00-003, Energy Laboratory, Massachusetts Institute of Technology, Cambridge, MA, October.

Westskog, Hege, (1996), "Market Power in a System of Tradeable CO2 Quotas," The Energy Journal 17(3):85-103.

Do Eco-Communication Strategies Reduce Energy Use and Emissions from Light Duty Vehicles?

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Abstract

The widespread use of eco-marketing and labeling strategies suggests they are perceived effective in promoting eco-conscious buying. However, some have been skeptical about the touted environmental and economic benefits of these programs. We present results from an ongoing study designed to test the effectiveness of a voluntary eco-communication program in the light-duty passenger vehicle market. Our results indicate consumers do value the benefits of greener vehicles but that the current state of eco-communication in this market is limited. We find that producers are reluctant to participate. It thus remains an open question whether a voluntary eco-communication program in the light-duty vehicle market will lead to an environmentally sustainable outcome.

Introduction

The environmental characteristics of products have become increasingly important to consumers. Firms have responded by placing information on existing products that highlight the product's environmental attributes and by introducing new, or redesigned, "green" products. Governments and non-governmental organizations have also responded by organizing, implementing, and verifying environmental labelling and marketing programs (hereafter, eco-information programs) that cover thousands of products in more than 20 countries [1]. Recently, the State of Maine has implemented the *Maine Clean Car Campaign* (hereafter the Campaign) - a cooperative effort of Maine's Department of Environmental Protection (DEP), the Maine Automobile Dealers Association (MADA) and the Natural Resources Council of Maine. The goal of the Campaign is to educate Mainers about the effects of vehicle air pollution and to inform them about differences in vehicle emissions. From a policy perspective, one aim of eco-information programs is to educate consumers about the environmental impacts of product consumption, thereby leading to a change in purchasing behavior, and ultimately, to a reduction in environmental impacts.

In the light-duty vehicle (cars, truck, minivans, SUVs) market, product regulation, while very effective in cases where consumers have no impact on outcomes, such as the elimination of lead in gasoline, are less effective when consumers can choose vehicles with different levels of environmental performance. This is seen most dramatically in the market shift from cars to light-duty trucks in the United States since 1975 when light-duty trucks had a market share of 21% to 2004 when their market share rose to 55% of all new passenger vehicles [2, Table 4.6]. This may well reflect that consumers' were unaware of the differences in the environmental performance of cars and trucks. Effective implementation and regulation of eco-information programs may allow customers to make choices that clearly reflect their environmental preferences while simultaneously achieving policy objectives (e.g., reductions in fossil fuel use and air emissions). Finding policy tools complementary to, or as a substitute for, command-and-control regulations is important. Indeed, the success of voluntary agreements, such as the Memorandum of Understanding between the Government of Canada and the Canadian automobile industry to limit GHG emissions from new vehicles, depends, in part, on the ability of vehicle manufacturers to sell consumers more fuel efficient vehicles [3].

Eco-communications programs may not achieve their objectives unless consumers are willing to pay for the underlying improvements in the production practices specified by the program. Earlier work has indicated that there is a portion of consumers who state they are willing to pay a premium for environmentally better vehicles. [4] [5] [6]. However, in addition to being willing to pay, consumers must also notice, understand and believe the information presented to them by the product manufacturer. Because the promise of improved production practices is impossible for most consumers to verify, the success of eco-labeling uniquely hinges on companies being able to credibly communicate to the consumer that some vehicles are environmentally better than others.

Although studies indicate a demand for 'greener' vehicles, no one has studied whether an eco-information program is effective in *altering* consumers' attitudes toward, or purchases of, environmentally preferred vehicles. It is, thus, an open question whether informed customer choice in the light-duty market will lead to these outcomes. Recent implementation of the Campaign provides an excellent opportunity to identify whether eco-information programs are effective. This study focuses on documenting the ability for eco-information to alter consumer attitudes because these are important antecedents to environmentally preferred behavior. We focus on the vehicle market because this market is of particular concern to policy makers. This concern arises in part from the fact that nationally, light-duty vehicles produce 62 percent of transportation CO2 emissions (including international bunker fuels). Combustion of fossil fuels to power transportation was the single largest source of greenhouse gas emissions in the U.S. economy in 1999 [7]. Light-duty vehicles are also responsible for 18 percent of nitrogen oxide (NOx), 45 percent of carbon monoxide (CO), and 26 percent of volatile organic compounds [7]. This is also true in Maine where on-road vehicles are the largest source of in-state created air pollution.

Design of the Maine Clean Car Campaign

There are two main parts to the Campaign. One part (eco-labeling) focuses on providing information to vehicle shoppers at the point-of-purchase (car and truck dealerships); although providing some educational benefit, the primary purpose of the eco-labeling is to provide information to improve consumers' ability to make cross-product comparisons. This dealer-based information consists of brochures explaining the Campaign, and placement of Clean Car stickers (Figure 1) in the window of new, environmentally better vehicles. The label indicates the vehicle:

- has a highway fuel economy rating of 30 miles per gallon or better; and
- is classified as a low emission vehicle by the U.S. Environmental Protection Agency.

Although the Maine Auto Dealers Association is an active partner in this Campaign, we should emphasize that active participation of individual dealers is solely voluntary. Indeed, one aspect of this research is to measure the level of participation among dealers, and the level of knowledge or awareness among sales people at the dealership.

The second part of the Campaign primarily focuses on educating Mainers about Maine's air quality,[1] its link to motor vehicles and to heighten awareness of the Campaign. This is important as research has shown that the credibility and effectiveness of an eco-labeling program is dependent upon consumer familiarity with the program [8].

The eco-marketing has several components. First, is the Campaign website (www.LEVforME.com) that provides detailed information about vehicles and their contribution to air quality problems. In addition, the Campaign uses newspaper and radio advertisements[2] that provide eco-information messages including information about the Campaign and the Campaign website. The eco-marketing portion of the Campaign started on January 31st and, except for the weeks of March 7, April 18 and May 30,[3] ran continuously until June 13th in several Maine newspapers and on radio stations of various formats (e.g., light rock, classic hits, and country music). Newspaper ads ran three times a week (Thursday, Friday and weekend editions) as banners (Figure 1) and were located on the third page of the front section. In total there were 20 different versions of the newspaper banner ads that where rotated within, and across weeks, to enhance repetition of messages across weeks while remaining 'fresh' (i.e., we did not run the same add on all three days of the week). All of the banner ads were designed to look similar, all carried a general environmental message related to vehicle emissions and all carried a representation of the eco-label. In aggregate, the 20 banner ads were displayed in the newspapers 153 times during the marketing treatment period.

Because one purpose of the research component of the project is to document the effects of the Campaign, the above eco-marketing program was only administered to one portion of the state. Hereafter, the portion of the state where the eco-marketing was administered is referred to as the treatment group and the remaining portion of the state is referred to as the control group.

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[1] Both in terms of criteria pollutants and global warming gases.

[2] The marketing materials (brochure, website, radio and newspaper advertisements) were designed with a Portland-based firm - BFT International LLC©.

[3] The weeks of March 7 and April 18 were school vacation weeks and May 30 was the week of Memorial Day holiday weekend.

Design of the University of Maine Study

The goal of the project is to determine whether the eco-labeling and marketing program had any impact on producer (vehicle dealers and sales personnel) knowledge and behaviors, and consumer knowledge, perceptions and behaviors. To determine impacts on producers we analyzed observational data collected from vehicle dealerships in Maine. To determine impacts on consumers we collected and analyzed two sources of data. First, we examined the level of activity being generated by the Campaign's website. Second, we analyzed changes in survey responses to a mail survey administered twice (before and after the marketing program) to independent samples of individuals residing in and out of the marketing treatment areas. Thus, our survey-based research design is quasi-experimental, with pre- and post-test measures from a treatment and a control group. This design helps isolate the impact of the eco-information program. Currently, we are almost through analyzing the survey data and are beginning the analysis of the market data; in turn, this paper will focus on only the survey results.

Literature Review

Economic theory suggests demand for a product or service is a function of a number of factors; one of these being the tastes and preferences of consumers. Traditionally, economists have been rather ill-equipped at incorporating tastes and preferences in their models (often proxied by socioeconomic characteristics). Yet, social psychologists have a rich literature focusing on what constitutes tastes and preferences.[4] This literature suggests a person's eco-behavior is positively influenced by their level of environmental involvement, perceived consumer effectiveness, and their faith in the eco-behavior of others. Barriers to environmentally friendly consumption include when individuals perceive that purchasing eco-products entails some increased inconvenience, cost or risk, or entails accepting a decrease in product quality. Teisl also finds that the amount of information can alter the perceived credibility of a label [10]. In general, adding information increases a label's credibility; however, adding some types of information could actually decrease it.

Methods

Observational study of vehicle dealers

To determine the level of dealer participation in the eco-marketing, we had several student employees attempt to visit all car and truck dealers who were members of the Maine Automobile Dealers Association (MADA) during the previous year. Visits to dealers were performed from June 24 to July 22, 2005; this was toward the end of the marketing treatment. Of the 134 eligible dealers, 105 were visited for a 78 percent visitation rate.

During a visit, the students would indicate to dealers that they were interested in purchasing a vehicle and were interested in an environmentally friendly vehicle. Students would mention the Maine Clean Car Campaign by name, and indicate they had learned of the program via radio or newspaper ads. They would then request additional information from the dealer including: a) looking for vehicles displaying the vehicle sticker, b) a brochure and c) where additional information regarding this program may be obtained (i.e. websites). Students would also enter the showroom to ascertain whether brochures or stickers were displayed in the showroom at each dealership. Additionally, students would look at vehicles on the lot which qualify for the program to confirm whether stickers were present on these vehicles.

For each visited dealership, students recorded whether the dealership displayed the Campaign's stickers on qualifying vehicles, and whether the Campaign brochure was available (displayed in the showroom or provided by a salesperson when requested). Students also recorded whether the

[4] See [9] for a more complete review of this literature.

salesperson knew about the Campaign's or DEP's websites where information about qualifying vehicles is listed. Finally, the students recorded qualitative information about the apparent level of knowledge exhibited by the salesperson.

Analysis of these data is along two fronts. First, we want to document the level of dealer participation in the Campaign to help us determine the relative importance of dealer-based eco-labeling versus non-dealer provided eco-marketing. Second, we want to analyze the factors that influence dealer participation in the Campaign, and the factors influencing sales personnel knowledge of the Campaign. In this paper we will use descriptive statistics to provide some analysis supporting the first objective.

Campaign website activity

All of the eco-marketing materials (brochures, newspaper and radio advertisements) related to the Campaign contained a website address. This website address was included for two reasons. First, previous research has indicated that the presence of a website address can increase the credibility of an eco-label [10]. Second, the website allowed us to reduce the level of detail presented in the marketing materials; this provided more interested consumers the ability to seek out more information while simultaneously reducing the potential for information overload occurring for less interested consumers.

The website (www.levforme.com) consists of a home/welcome page, and several content and ancillary pages. One content page ("What's the problem?") presents information about the environmental problem and its link to vehicle emissions whereas another page ("What can you do?) presents suggestions on how to be a more environmentally conscious driver/vehicle owner. Others pages are devoted to presenting background information about: the project partners ("Who are we?"), the sticker (eco-labeling) program, the components of the eco-marketing program ("See & hear the campaign") and more academic reports generated by the project ("Want more info?").

To determine the quantity and quality of website activity we collected the following daily web statistics; number of unique visitors to the site, number of hits, length of visit and pages visited. We use descriptive statistics to analyze these data.

Consumer mail survey

In May of 2004 and 2005, approximately 1.4 million vehicle registration records were obtained from the Maine Bureau of Motor Vehicles; the records (our sampling frame) represent everyone who registered a vehicle in Maine within the previous 12 months. A random sample of approximately 2,000 was generated each year from the frame with approximately 800 records removed because they were inappropriate or contained incomplete information. For example, records were rejected if the: primary address was outside the state, vehicle was listed as homemade, registration was for a non-passenger vehicle (e.g., utility trailers) or records did not have a valid vehicle identification number. Multiple registrations were also removed.

Between June and August of each year, we administered a mail survey to final random samples of 1,148 and 1,163 (2004 and 2005 surveys respectively) Maine adults who had registered vehicles in Maine. In total, 620 Maine residents responded to the 2004 survey and 691 responded to the 2005 survey, for responses rates of 60 and 64 percent, respectively. In general, our respondents are similar to the characteristics of the Maine adult population as measured by the recent U.S census, except in terms of gender. Although our survey respondents are more likely to be males, relative to the U.S. census, the proportion of males correctly reflects the underlying percent of males in the vehicle registration data.

The survey instrument consisted of seven sections with forty-one questions. Sections I and II solicited respondents' opinions on air quality in Maine, the relationship between motor vehicles and air pollution, and environmental protection in general. Section III asked respondents about their current vehicle, including the type of vehicle and the importance of various attributes considered during the purchase decision. Section IV respondents were asked about their search and use of environmental information in the vehicle purchase decision. Sections V and VI incorporated an experimental label test and a vehicle choice experiment, respectively. The final section of the survey, Section VII, collected demographic characteristics.

To evaluate the success of the eco-marketing program we need to examine whether the Campaign altered people's environmentally-related knowledge and perceptions as these have been shown to be important antecedents to supporting eco-behaviors. To determine whether the eco-marketing program affected these psychological variables we estimated a series of models which differ in their dependent variables but were of the general form of:

$$DEP = \alpha + \beta_1 YEAR + \beta_2 MKT + \beta_3 GEN + \beta_4 AGE + \beta_5 ENV + \beta_6 ED + \beta_7 INC + \Sigma_k \beta_{8k} REC_k + \beta_9 NOREC$$

where DEP denotes the dependent variable which varies across equations (see Table 1 for definitions of all variables). The dependent variables included one variable to measure exposure to the Campaign information (SEE), one behavioral variable (SEARCH) and various perceptual/psychological measures (WANT, DLR, IMP, CONC, AQUAL, LSTYLE, 2HARD, MOST, WTP, LAWS, TRST, ALLS, LPERF, and COST). SEE is our most basic measure of the campaign's success as an information program cannot succeed unless it is first noticed and recognized. WANT, DLR and IMP denote whether respondents want information that helps them identify vehicles that produce less pollution when driven, find auto dealers are good at providing this type of information and measure the importance a respondent places on eco-label information. We asked a series of questions (CONC, AQUAL) to determine Mainers' opinions of Maine's current level of air quality;[5] increased levels of concern should indicate an increased likelihood for the Campaign to succeed.

The variables of LSTYLE, 2HARD, MOST, WTP, LAWS, TRST, LPERF, and COST are coded from respondents reactions to a set of perceptual statements aimed at measuring their general perceptions regarding their personal environmental impact, others' willingness to work to improve the environment, whether science or the state can effectively reduce air pollution. Responses to the first set of questions (LSTYLE, 2HARD) provide information about whether individuals see themselves as being able to improve the environment through the choices they make. Presumably, individuals who see their choices as having an environmental impact are more likely to take notice of Maine's Clean Car Campaign. The second set of questions (MOST, WTP) is meant to measure individuals' perceptions of others' level of environmental involvement; responses to these questions have several possible interpretations. Individuals who perceive that other people are environmentally involved may feel increased pressure to act similarly (a 'peer-pressure' effect). Alternatively, these individuals may think that, since others are doing their share for the environment, they do not need to do anything to improve the environment (a 'free-rider' effect). The third set of questions (LAWS, TRST) is meant to measure individuals' perceptions about whether they think the state is capable of improving or safeguarding environment quality. The last set of questions (LPERF, COST) was aimed at seeing what people view as some of the perceived tradeoffs when buying a greener vehicle. Individuals who see greener vehicles as being inferior substitutes are less likely to respond positively to the Campaign.

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[5] The question only asked about general air quality concerns and did not differentiate between criteria pollutants and global warming gases. However, other research indicates that vehicle buying decisions are driven more by a concern with global warming gases [9].

ALLS measures whether respondents think all vehicles pollute about that same when driven; eco-labeling, dependent upon the idea there are significant eco-differences across products, should be less important to individuals holding priors that there are no environmental differences across vehicles.

Results

Participation by vehicle dealers

Of the 89 dealerships, only 10 (11 percent participation) displayed the Campaign sticker on its vehicles and only 10 (11 percent) had the Campaign brochure available; only four dealerships (four percent) provided both the labels and the brochure. Sales personnel knowledge of the Campaign or DEP websites was also low; only two (two percent) of the contacted salespeople knew about the Campaign website and only four (four percent) knew about the DEP website. In terms of a general knowledge on the environmental characteristics of vehicles, sales people did rather better; 22 (25 percent) exhibited some awareness or knowledge of the Campaign and 13 (15 percent) knew about vehicles meeting California's emission standards. In terms of a willingness to assist the customer, five of the contacted salespeople used their computer to link to the Campaign or DEP websites.

Overall, we find these results from dealerships particularly disappointing. Perhaps, however, we should not be surprised. In our discussions with the Maine Auto Dealers Association we stressed that all full-line vehicle manufacturers would have some vehicles that qualified for the label. Hence the label could be used as a positive selling point to consumers who value the environmental performance of their vehicles. This positive approach presumes that dealers are indifferent to which vehicles they sell. This may not be borne out in practice. For purposes of inventory management or because of differences in per vehicle profits, dealers may prefer to sell low scoring vehicles (or not draw attention to the fact that some of their vehicles score low). This may lead dealers to consciously choose not participate in the sticker program and not educate their sales staff. Whatever the cause, the low-level awareness of sales people at the dealerships in the sticker program shows that a voluntary approach to vehicle labeling may not be effective in promoting a positive sales approach to clean vehicles. It would be interesting to test whether a mandatory approach to vehicle labeling would be more effective in selling environmentally friendly vehicles.

Campaign website activity

Descriptive statistics indicate there was a relatively strong increase in website activity once the newspaper and radio advertisements began at the end of January (Figure 2). Before the Campaign there were only 10 visitors/122 hits during the month of January. After the Campaign, this rose to 85 visitors/3,000 hits (during February); which settled to an average of 150 visitors per month and 2,200 hits per month over the next few months. Initially there were a greater numbers of hits to the site relative to the number of visitors. This difference declined after the first few months of the campaign. This would seem to indicate that initially, there were a greater number of visitors making repeated hits to the site. Interestingly, the level of website activity was maintained for about six months after which it began to increase (except for a decline in activity during the summer of 2006) – this was occurring months *after* cessation of radio and newspaper advertising. Apparently, the website continued to attract attention and it is currently unclear why this occurred. It could be that 'word-of-mouth' advertising (either by previous website users or by sales personnel at dealerships) began generating its own stream of new visitors. It could also be an artifact of how internet search engines operate; as sites generate more hits they move up the ranking of most search engines – leading to a future increased level of traffic. We are continuing to examine the data to try and explain what caused this phenomenon;

however, we have noticed that a percentage of our visitors are from outside the state (in fact, some of our visitors are from outside the US). This may indicate that news of the website may be spreading from among eco-conscious web-users.

The number of visitors/hits seems to indicate a positive eco-marketing effect; however, the data on length of visit (average visit length was 4 ½ minutes) indicates that most visits lasted for less than half a minute (Table 2). This may indicate that many visitors were simply searching for some quick information to establish the legitimacy of the campaign or link to another site (e.g., using the website to link to Maine DEP site which lists the vehicles qualifying for the eco-label). Alternatively, it could mean that many visitors only stumbled onto the site by accident and then left immediately. We do not know which explanation is more likely, however, data on which pages were being visited suggests that many visitors were indeed searching for some more information. Most visitors went beyond the welcome/home page of the site and visited other pages. The most popular page (54 percent of visits) was the one presenting information about the environmental problem and its link to vehicle emissions ("What's the problem?"). Three pages ("What can you do?, "Who are we?" and "See & hear the campaign") were tied for the second most popular; each being visited by 11 percent of visitors.

Regression analysis: Changes in consumer perceptions and norms

We begin this section by presenting the results from the regressions, starting with some general observations of the impacts of individual characteristics, followed by a discussion focusing on the impact of the marketing treatment. Although we present the parameter estimates related to an individual's participation in outdoor recreation activities due to paper length we will not discuss these results.

<u>Respondents' perceptions and experiences with environmental information</u>

Males are less likely to want environmental information about vehicles; probably because they see this information as being less important (Table 3). In contrast, older individuals are more likely to search for information about how much pollution the vehicle generate and desire this information. More educated individuals and environmentalists are the most positive about the value of environmental information; they both tend to see the information as important and desirable, and search for this type of information when they are looking to buy or lease a vehicle.

<u>Respondents' perceptions of the environmental problem</u>

Males are generally less concerned about Maine's air quality relative to females (Table 4). They are less likely to think that their lifestyle has an impact on the environment and are more likely to think that it's too hard for someone like them to do much about the environment. However, males tend to think that most others are doing their part to protect the environment. Interestingly, males tend to think that current air pollution laws are strong enough but are less likely to trust the state government to protect the environment.

Older individuals are more concerned about Maine's air quality and tend to rate Maine's current air quality as poor. Similar to males, older respondents are more likely to think that its too hard for someone like them to do much about the environment and that most other people are willing to pay higher prices to protect the environment. Unlike males, older respondents tend to think that current air pollution laws are too weak but they are more likely to trust the state government to protect the environment.

Individuals who claim to be environmentalists are more concerned about Maine's air quality and tend to rate Maine's current air quality as poor. They are more likely to think that their lifestyle has an impact on the environment and less likely to think that it's too hard for someone like them to do much about the environment. Interestingly they have a more negative view of others' environmental behaviors. Not surprisingly they tend to think that current air pollution laws are too weak.

More educated individuals are generally less concerned with Maine's air quality. They are less likely to think that it's too hard for someone like them to do much about the environment. More educated individuals have a more negative view of others' and the state's environmental behaviors.

Respondents' perceptions of vehicles

Males, older and more educated individuals are less likely to hold the view that all vehicles pollute about the same when driven (Table 5). Older individuals tend to think that environmentally better vehicles suffer from poorer performance but are less likely to think that these vehicles are more expensive. Environmentalists are more positive about environmentally better vehicles; they are less likely to think that these vehicles suffer from poor performance or are more expensive.

Impacts of the marketing treatment

In each equation, the intercept parameters measure the average baseline (2004) level of the dependent variable (both for the treatment and control groups). The parameter on YEAR measures the change in the average responses of individuals who were not exposed to the eco-marketing campaign (the control group) during 2005 whereas the parameter on MRKT provides similar measures for individuals who were exposed to the marketing (treatment group) in 2005.Thus, to measure whether there was any impact on consumer perceptions and experiences we need to examine both the parameters on YEAR and MRKT (Tables 3, 4 and 5) and test whether there are significant differences between these parameters (Table 6). We use linear hypotheses tests to indicate whether the parameters on YEAR and MRKT are statistically different from each other.

In total we have 16 equations and we find that responses from individuals exposed to the marketing treatment are in the desired direction in 15 of them, and significant in five of the equations. Importantly, individuals exposed to the marketing treatment are significantly more likely to recognize the Campaign eco-label, a minimum requirement of the eco-marketing program. Additionally, we find that individuals exposed to the eco-marketing have a more pessimistic view of Maine's current level of air quality and are more likely to view current air pollution control laws as weak. Movements in these variables should increase the effectiveness of the eco-labeling portion of Campaign.

We also find that individuals exposed to the eco-marketing place a greater faith on the state's abilities to protect Maine's environment and in other people's willingness to pay for environmental protection. This is consistent with Bamberg's [11] contention that normative expectations of others may be a positive factor in an individual's behavior and by Gould and Golob's [6] work, where they indicate the positive behavior of others influences drivers' sense of personal responsibility for vehicle air pollution.

That the other "correct" effects of the marketing program are insignificant does not necessarily indicate a general ineffectiveness of eco-marketing programs. Analyzing responses to any marketing program is similar to analyzing a dose-response function. The potential magnitude of the effect is related to the size of the 'dose'. On several fronts, our marketing effort was of a relatively low dose: the entire program cost less than $125,000, did not use other available media (e.g., television) and ran for only about four and a half months. In addition, the low level of participation among dealers and the low level of awareness and knowledge among sales personnel likely limited the overall impact of program. Finally, some of our perceptual measures are more general and are likely to be ones that are less amenable to a significant marketing effect. The fact that we find correct impacts in all but one equation suggests a stronger marketing effort would be associated with more significant, positive responses.

Conclusions

The flow of information among market participants can play a critical role in the efficient operation of markets. In a broad sense, eco-information programs have the ability to convert a market in which all goods feature an attribute that consumers can't observe, or may not know about, into one in which consumers can or do. From a policy perspective, these programs allow consumers to make choices which match personal preferences and may provide information that actually changes people's preferences. From a business perspective, these programs may allow firms using particular techniques to gain market share.

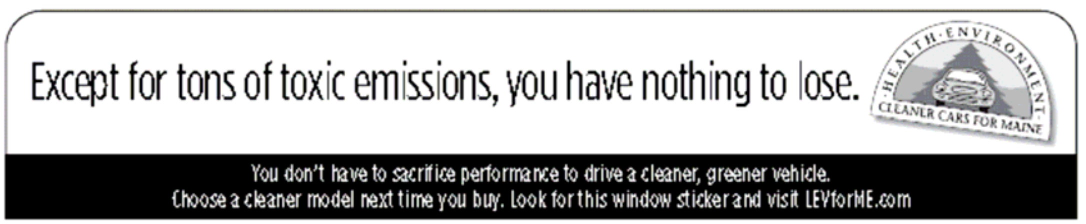
The results indicate the potential importance of well-designed eco-labeling and marketing strategies. The ability of eco-marketing information to alter the underlying psychological factors (both social and personal norms) shown to be important in eco-buying behavior suggests a strong (or perhaps new) role for the long-run provision of information through eco-marketing or eco-education programs. Providing eco-labels without an eco-marketing program to alter consumers' prior perceptions (especially when they are incorrect) may lead to less effective programs.

The reverse may also be true; providing eco-marketing without a strong eco-labeling component can also limit the programs effectiveness. For example, one of the prime messages of the eco-marketing portion of the Campaign was that vehicles are significantly different in their environmental characteristics. However, more than half of the respondents (approximately 60 percent) stated they thought most vehicles pollute about the same; in stark contrast to the reality of car and truck pollution. The success of an eco-information strategy in the vehicle market is contingent on people understanding that the choices they make in buying a vehicle can have significantly different impacts on the amount of air pollution generated. Yet even after exposure to the eco-marketing campaign, respondents have an imperfect appreciation for the large differences in the amount of air pollution produced by different types of vehicles. This continuing misperception is probably due to the lack of vehicle-specific emissions information (eco-labeling) present in the market. According to our survey results, almost half of our respondents (47 percent) visited a car or truck dealership within the last year; however, most of these consumers were never exposed to vehicle-specific emissions information because only a minority of dealerships participated in the Campaign. Presumably, if more dealers participated (or were made to participate) then consumers would be more familiar with the eco-labels and be more cognizant of the differences between vehicles.

Figure 1. Example of the Maine's Clean Car sticker (eco-label), placed on all new vehicle models meeting the Clean Car Program standards, and an example of a newspaper banner advertisement.

Eco-label

Newspaper banner ad



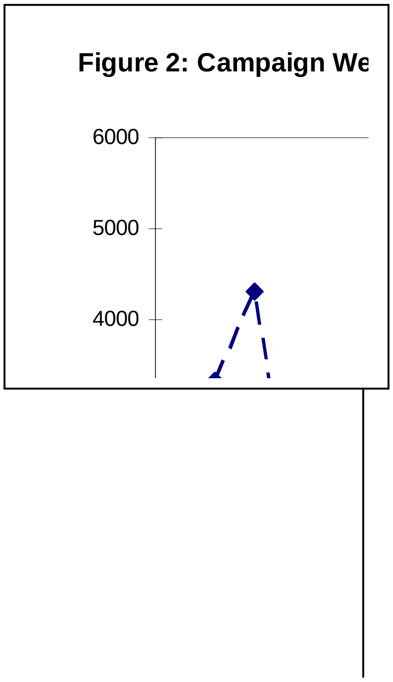


Figure 2: Campaign We
6000
5000
4000

Table 1. Variable names and descriptions

Name	Description
Dependent variables	
SEE	Coded 1 if respondent saw the campaign's eco-label; 0 otherwise
SEARCH	Coded 1 if respondent stated before buying or leasing a new vehicle they searched for information about how much pollution the vehicle produces
WANT	Coded 1 if respondent wants information helping them identify vehicles that produce less pollution; 0 otherwise
DLR	Coded 1 if individuals find auto dealers are good at providing information about how much air pollution a vehicle makes; 0 otherwise
IMP	The importance a respondent places on eco-label information (1 = 'not at all important' to 5 = 'very important')
CONC	The individual's concern over the amount of air pollution in Maine (1 = 'not at all concerned' to 5 = 'very concerned')
AQUAL	The individual's rating of Maine's air quality (1 = 'very bad' to 5 = 'very good')
LSTYLE	Respondent agrees[a] with the statement: My lifestyle can have an impact on the environment
2HARD	Respondent agrees[a] with: It is too hard for someone like me to do much about the environment
MOST	Respondent agrees[a] with: Most people do their part to protect the environment
WTP	Respondent agrees[a] with: Most people are willing to pay higher prices to protect the environment
LAWS	Respondent agrees[a] with: Air pollution laws are already strong enough
TRST	Respondent agrees[a] with: I trust the state government to protect Maine's environment
LPREF	Respondent agrees[a] with: Vehicles that produce less pollution have lower performance
COST	Respondent agrees[a] with: Vehicles that produce less pollution are more expensive
ALLS	Coded 1 if respondent thinks all personal vehicles pollute about the same when driven; 0 otherwise
Independent variables	
YEAR	Coded 1 if data was collected during 2005; 0 if during 2004
MKT	Coded 1 if respondent lived in the eco-marketing treatment area; 0 otherwise
GEN	Coded 1 if respondents is male; 0 if female
AGE	The respondent's age in years
ENV	Coded 1 if the respondent belonged to an environmental organization; 0 otherwise
ED	The respondent's education level in years
INC	The respondent's income in dollars
NOREC	Coded 1 if respondent did no outdoor recreation in the past year; 0 otherwise
BIKE	Coded 1 if respondent mountain or road biked; 0 otherwise
WATCH	Coded 1 if respondent wildlife watching; 0 otherwise
SNOW	Coded 1 if respondent snowmobiling; 0 otherwise
PHOTO	Coded 1 if respondent participated nature photography; 0 otherwise
BOAT	Coded 1 if respondent boated/canoed; 0 otherwise
HUNT	Coded 1 if respondent hunted; 0 otherwise
ATV	Coded 1 if respondent participated in ATV or dirt biking; 0 otherwise

a 1 = 'strongly disagree to 5 = 'strongly agree'

Table 2. Distribution of length of visit to the website; in minutes

Length of visit	Percent
Less than half a minute	78
Thirty seconds to two minutes	5
Two to five minutes	4
Five to fifteen minutes	5
Fifteen to thirty minutes	3
Thirty minutes to an hour	3
Greater than an hour	2

Table 3. Regression results: Respondent's perceptions and experiences with environmental information.[a]

	SEE	SEARCH	WANT	DLR	IMP
Intercept	-4.791***	-4.130***	-0.380	-3.316***	-1.783***
Intercept					-0.601*
Intercept					1.475***
Intercept					2.431***
YEAR	-0.806*	-0.130	-0.112	-0.067	-0.030
MKT	1.643***	0.179	0.071	0.284	0.033
GEN	-0.054	0.186	-0.482***	0.201	-0.920***
AGE	0.0043	0.028***	-0.002	0.023***	0.0035
ENV	0.427	0.647***	0.674***	0.183	0.976***
ED	0.041	0.045*	0.112***	-0.001	0.036*
INC	2.6E-6	5.2E-6**	-9.8E-7	2.7E-6	-4.4E-7
NOREC	-1.259*	-0.433**	0.135	-0.264	-0.099
BIKE	-0.620	0.160	0.238	0.274	0.261**
WATCH	0.405	0.142	0.522***	-0.248	0.349***
SNOW	0.561	-0.158	-0.240	-0.174	-0.340**
PHOTO	-0.906**	0.374**	0.144	-0.316	0.041
BOAT	0.010	0.228*	-0.077	0.404***	-0.072
HUNT	0.894***	-0.282	-0.389***	0.300*	-0.364***
ATV	-0.225	-0.287	0.052	-0.280	-0.097

[a] * denotes significant at the 10 percent level;** denotes significant at the five percent level;*** denotes significant at the one percent level

Table 4. Regression results: Respondent perceptions of: the environmental problem, their personal involvement in protecting the environment, others peoples participation in environmentally friendly behaviors and the state's ability to protect the environment.[a]

	CONC	AQUAL	LSTYLE	2HARD	MOST	WTP	LAWS	TRST
Intercept	-1.422***	-1.666***	-0.724**	-2.556***	-2.918***	-4.401***	-1.622***	-2.881***
Intercept	-0.371	0.444	1.625***	-0.693**	-0.625*	-1.443***	-0.215	-0.798**
Intercept	1.938***	2.992***	2.728***	0.175	0.255	-0.692**	0.831**	0.200
Intercept	2.865***	5.482***	4.014***	2.258***	2.348***	1.198***	2.255***	1.688***
YEAR	-0.042	0.440***	0.215*	0.0064	-0.153	-0.287***	0.162	-0.163
MKT	0.134	-0.327***	-0.009	-0.014	0.032	0.061	-0.250**	0.261**
GEN	-0.691***	0.086	-0.294***	0.292***	0.529***	0.029	0.320***	-0.275***
AGE	0.021***	-0.014***	-0.013***	0.0046	0.005	0.018***	-0.009***	0.0098***
ENV	0.615***	-0.381***	0.895***	-0.503***	-0.268*	-0.085	-0.595***	0.164
ED	-0.052***	-0.008	0.0092	-0.048**	-0.049**	-0.041**	-0.038**	-0.058***
INC	1.7E-6	6.1E-6***	5.7E-6***	-6.2E-6***	-4.2E-6**	6.9E-6***	1.9E-6	-9.0E-7
NOREC	-0.236*	0.093	-0.492***	0.629***	0.324**	0.303**	-0.032	0.433***
BIKE	0.356***	-0.214*	-0.022	-0.173	-0.301**	0.102	-0.231**	-0.185
WATCH	0.310***	-0.315***	0.273	0.341***	0.069	0.142	-0.362***	-0.280***
SNOW	-0.407***	0.779***	-0.003	-0.435***	0.006	0.060	0.386***	-0.0034
PHOTO	0.245*	-0.048	0.092	-0.418**	-0.195	0.219*	-0.126	-0.459***
BOAT	0.123	0.194*	0.255**	-0.280***	-0.150	-0.108	-0.184*	-0.070
HUNT	0.225*	0.087	-0.400***	0.358***	-0.052	-0.245**	0.249**	0.194*
ATV	0.034	-0.038	-0.446***	0.349**	0.083	0.116	0.127	0.214

[a] * denotes significant at the 10 percent level;** denotes significant at the five percent level;*** denotes significant at the one percent level

Table 5. Regression results: Respondent perceptions of vehicles.[a]

	ALLS	LPERF	COST
Intercept	2.120***	-2.821***	-0.363
Intercept		-1.010***	1.631***
Intercept		-0.040	2.496***
Intercept		1.478***	3.825***
YEAR	0.031	-0.189*	0.110
MKT	-0.052	-0.011	-0.016
GEN	-0.319***	0.115	-0.051
AGE	-0.008**	0.006*	-0.012***
ENV	-0.208	-0.427***	-0.253*
ED	-0.097***	-0.009	-0.023
INC	2.2E-6	2.0E-6	-2.4E-6
NOREC	-0.156	0.023	-0.143
BIKE	-0.152	-0.293**	-0.195*
WATCH	-0.297***	-0.049	0.171*
SNOW	-0.275*	0.323**	0.050
PHOTO	0.492***	0.058	-0.320**
BOAT	0.108	0.106	-0.092
HUNT	0.470***	0.201*	-0.046
ATV	0.603***	-0.152	-0.284**

[a] * denotes significant at the 10 percent level;** denotes significant at the five percent level;*** denotes significant at the one percent level

Table 6. Summary of marketing impacts[a]

Equation	Desired sign	Is sign met?	χ^2	p
Respondent perceptions and experience with environmental information				
SEE	+	Yes	7.744	0.005
SEARCH	+	Yes	1.096	0.295
LIKE	+	Yes	0.501	0.479
DLR	+	Yes	1.243	0.265
IMP	+	Yes	0.090	0.762
Respondent general environmental perceptions				
CONC	+	Yes	0.723	0.395
AQUAL	-	Yes	13.202	0.000
LSTYLE	+	No	1.093	0.296
2HARD	-	Yes	0.010	0.920
MOST	+	Yes	0.802	0.370
WTP	+	Yes	2.869	0.090
LAWS	-	Yes	4.122	0.042
TRST	+	Yes	4.281	0.038
Respondent perceptions of vehicles				
ALLS	-	Yes	0.128	0.720
LPERF	-	Yes	0.774	0.379
COST	-	Yes	0.380	0.538

a Degrees of freedom for all chi-square tests is equal to one

References

1. U.S. Environmental Protection Agency, Determinants of Effectiveness for Environmental Certification and Labeling Programs (1994).
2. S. C. Davis and S. W. Diegel, Transportation Energy Data Book Vol. 25 Oak Ridge National Laboratory (2006).
3. Natural Resources Canada, Memorandum of Understanding Between the Government of Canada and the Canadian Automotive Industry Respecting Automobile Greenhouse Gas Emissions (2005).
4. C. L. Noblet, Factors Affecting Consumer Assessment of Eco-Labeled Traditional Fuel Passenger Vehicles, *in* "Department of Resource Economics and Policy", University of Maine, Orono (2005).
5. D. Brownstone, D. S. Bunch, T. F. Golob and W. Ren, A Vehicle Transactions Choice Model for Use in Forecasting Demand for Alternative-Fuel Vehicles, *Research in Transportation Economics* (1996), **4**, 87-129.
6. J. Gould and T. F. Golob, Clean Air Forever? A Longitudinal Analysis of Opinions About Air Pollution and Electric Vehicles, *Transportation Research - D* (1998), **3**, 157-169.
7. USEPA (US Environmental Protection Agency), Inventory of US Greenhouse Gas Emissions and Sinks: 1990-2002 (2004).
8. J. Thøgersen, Promoting Green Consumer Behavior With Eco-Labels, *in* "National Academy of Sciences/National Research Council Workshop on Education, Information and Voluntary Measures in Environmental Protection", Washington, D.C. (2000).
9. M. F. Teisl, C. L. Noblet and J. Rubin, The Design of an Eco-Marketing and Labelling Programme for Vehicles in Maine, *in* "New Frontiers in Environmental and Social Labeling", Springer (forthcoming).
10. M. F. Teisl, What We May Have is a Failure to Communicate: Labeling Environmentally Certified Forest Products, *Forest Science* (2003), **49**.
11. Bamberg S., How Does Environmental Concern Influence Specific Environmentally Related Behaviors? A New Answer to an Old Question., *Journal of Environmental Psychology* (2003), **23**, 21-32.

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VEHICLE CHOICES, MILES DRIVEN AND POLLUTION POLICIES

Ye Feng
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Vehicle Choices, Miles Driven, and Pollution Policies
Ye Feng, Don Fullerton and Li Gan
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ABSTRACT

Mobile sources contribute large percentages of each pollutant, but technology is not yet available to measure and tax emissions from each vehicle. We build a behavioral model of household choices about vehicles and miles traveled. The ideal-but-unavailable emissions tax would encourage drivers to abate emissions through many behaviors, some of which involve market transactions that can be observed for feasible market incentives (such as a gas tax, subsidy to new cars, or tax by vehicle type). Our model can calculate behavioral effects of each such price and thus calculate car choices, miles, and emissions.

A nested logit structure is used to model discrete choices among different vehicle bundles. We also consider continuous choices of miles driven and the age of each vehicle. We propose a consistent estimation method for both discrete and continuous demands in one step, to capture the interactive effects of simultaneous decisions. Results are compared with those of the traditional sequential estimation procedure.

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The standard case for market-based incentives requires a tax or price on each unit of emissions. Each form of abatement is then pursued until the marginal cost of reducing pollution matches the tax per unit of pollution, and the resulting combination of abatement technologies minimizes social costs (Pigou, 1920). For vehicles, a tax on emissions could induce drivers to: (1) buy a newer, cleaner car, (2) buy a smaller, more fuel efficient car, (3) fix their broken pollution control equipment, (4) buy cleaner gasoline, (5) drive less, (6) drive less aggressively, and (7) avoid cold start-ups.[1] Moreover, economic efficiency requires different combinations of these methods for different consumers: some lose little by switching to a smaller car, some could easily walk, and some just pay the tax.

Yet the technology is not available to measure each car's emissions in a reliable and cost-effective manner. On-board diagnostic equipment is imperfect, and it is costly to retrofit millions of vehicles (Harrington and McConnell, 2003). Remote sensing is less expensive and has been used to identify high-polluting vehicles, but it cannot measure emissions clearly enough to tax each car.[2] Moreover, vehicle emissions are important. In 2001, vehicles in the U.S. contributed 27 percent of volatile organic compounds (VOC), 37 percent of nitrogen oxides (NO_x), and 66 percent of carbon monoxide (CO) emissions.[3]

For these reasons, vehicle emission policies have relied almost solely on mandates: refineries must make clean gasoline, and new cars must meet required emission standards.[4] These command and control (CAC) policies miss the opportunity to reduce social costs by harnessing individual incentives, however, as the mandated combination of abatement methods is unlikely to match the combination that households would choose if faced with a tax on emissions. In fact, the cost of abatement using such mandates can be several times the minimum cost achieved by using an emissions tax (Newell and Stavins, 2003).

While the inability to measure emissions may preclude a vehicle emissions tax, it does not preclude any use of incentives. Those who sell new or used cars or light-trucks

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[1] Heeb et al (2003) find that cold start emissions rates (in g/km traveled) exceed stabilized emissions rates by a factor of two to five, depending on the pollutant. Sierra Research (1994) finds that a car driven aggressively has carbon monoxide emissions that are almost 20 times higher than when driven normally.

[2] See Sierra Research (1994). Remote sensing in Texas (http://www.tnrcc.state.tx.us/air/ms/vim.html#im3) and Albuquerque NM (http://www.cabq.gov/aircare/rst.html) is used in 2005 to identify polluting vehicles.

[3] See http://www.bts.gov/publications/transportation_statistics_annual_report/2004/. We focus on local pollutants, where emission rates depend on car characteristics. In contrast, CO_2 is linked directly to gas use.

[4] In the U.S., new cars face emission standards of .254 grams/km of HC's, 2.11 grams/km of CO, and .248 grams/km of NO_x. Light trucks face a variety of weaker standards, but all are scheduled to become more stringent. These figures pertain to a test in the U.S. with a cold start-up phase, a transient phase at different speeds, and a hot start phase, for a total distance of 18 km at an average speed of 34 km/h.

can collect tax on vehicle characteristics that are associated with emissions, or provide subsidy for vehicles with low emissions. Most states charge annual registration fees that can be made to depend on vehicle characteristics. Such policies might reduce emission rates, while changes in the gasoline tax can reduce miles driven.[5]

What vehicle characteristics or behaviors should be targeted by a tax or subsidy? How would consumers react to those new incentive instruments? How much would each tax reduce emissions? To address these questions, we build a general purpose model of discrete choices by households about how many cars to own and what types of cars to own, plus continuous choices about how far to drive. In our model, we embrace individual heterogeneity. We estimate all decisions simultaneously, and we use the estimated parameters to predict the effects of certain price changes on choices and on emissions.

Several existing papers explore market incentives that could be used in place of a tax on emissions.[6] In addition, several papers estimate models of the discrete choice among vehicle bundles (including number, size, and age categories).[7] Some models estimate the demand for gasoline or for vehicle miles traveled (*VMT*) as functions of price and income (as reviewed in Harrington and McConnell, 2003). As well, we note that other models predict emissions.[8] A major contribution of our research, then, is to include all such choices simultaneously. In general, we capture the effect of any price change on each household's choices about the number of vehicles to buy, the type and age of each, the consequent emissions rates, miles driven, and the consequent total emissions.

In a two-step procedure, Dubin and McFadden (1984) estimate a discrete choice model (for household appliances) and use the predicted shares to correct for endogeneity in the estimation of a continuous choice (usage hours). Others extend this model to the discrete choice among vehicle bundles and a continuous choice of miles (e.g. Goldberg, 1998, and West, 2004). Yet, a single set of parameters appear both in the indirect utility

[5] A new higher gas tax may be politically unlikely, yet it is still worth studying to know its power as an emissions-reduction tool. And even if governments are unlikely to use tax dollars to pay for the various subsidies we study here, these incentives might instead be provided to drivers by private companies that want to purchase "offsets" – reductions in vehicle emissions to offset their increases from stationary sources. For all of these reasons, we find it important to study specific incentives to drivers.

[6] For examples, see Eskeland and Devarajan (1996), Innes (1996), Kohn (1996), Train et al (1997), Plaut (1998), Sevigny (1998), and Fullerton and West (2000, 2002).

[7] See McFadden (1979), Mannering and Winston (1985), Train (1986), Brownstone et al (1996), Goldberg (1998), Brownstone and Train (1999), West (2004), and other papers reviewed in McFadden (2001).

[8] For example, the U.S. Environmental Protection Agency (U.S. EPA, 1998, p.3-68) discusses the use of EPA's MOBILE5a model or California's EMFAC7F model.

function used to estimate discrete choices and in continuous demands. Using this sequential procedure, the estimated parameters of the continuous demand are not constrained to match the same parameters in the estimated discrete choice model.

Relative to this literature, we make a number of contributions. First, we capture the simultaneity of these decisions by proposing a method for consistent estimation of both discrete and continuous choices in one step, yielding a single set of parameters. In other words, whereas the Dubin-McFadden method corrects for selection of vehicle on the choice of miles, our simultaneous procedure also allows for heterogeneity in actual fuel demand to affect the choice of vehicle.[9] Second, we allow for two continuous choices of miles – in each vehicle of a two-vehicle household. These choices are bundle-specific.[10] Third, we allow for an additional continuous choice of the age of each vehicle. Fourth, we use the estimated parameters not only to predict changes in choices about vehicles and miles, but also how those choices affect emissions.[11]

For several reasons, we deviate from discrete vehicle types used in prior literature (including age and size categories). First, we have no need to model the choice among hundreds of vehicle types, as in prior studies of manufacturer product differentiation, since all cars in a given year are made to a single emission rate standard. Second, a different, weaker emission standard has applied to "sports utility vehicles" (SUV, for short, but defined here to include all light trucks and vans). Emission rules for new vehicles do not depend on engine size. We therefore model the choice between car and SUV, rather than engine size. Even for older vehicles, when we use data described below in separate regressions for cars and SUV's, we find that engine size is not an important determinant of emission rates. Third, those regressions find that vehicle age is very important for emission rates. We wish not to lose information by aggregation into finite age categories (e.g. new

[9] Hanemann (1984) proposes a method to estimate these demands simultaneously, but his method does not consider unobserved individual heterogeneity – a key factor in the Dubin-McFadden model. Our model captures the individual unobserved heterogeneity. Bento et al (2005) and Bhat (2005) are also working on models with simultaneous discrete and continuous choices.

[10] With a higher price of gas, some households might drive fewer miles in their SUV and *more* in their car. We do not estimate separately the miles in each vehicle, but we do estimate a change for the (Car, SUV) bundle that can differ from the (Car, Car) bundle. Other papers have estimated substitution between vehicles within the family, but they treat the vehicles as given rather than chosen. Greene and Hu (1985) find that this kind of substitution occurs to a large extent in some households, while Sevigny (1998) finds small effects.

[11] Our household responses represent market outcomes only if supply curves were horizontal. The simulation of a change in the price of getting a car that is one year newer can be interpreted as a new local tax or subsidy in a small open jurisdiction that can import more of those newer cars at a constant price. However, our demand system could be combined with some other estimates of supply to calculate equilibrium outcomes.

vs. old). Age is a continuous variable, and the choice of vehicle age is a continuous demand that affects emissions.[12] If a household in our model chooses to own two vehicles, then it has four continuous choices: age of each vehicle and miles to drive each vehicle.[13]

Age is normally measured in years, of course, but our model requires a price that does not depend on the amount demanded. The price of age is not linear, because owning a brand-new car costs more depreciation per year than owning an old car. Instead of using age in years, we therefore construct a continuous choice variable called "*Wear*" that measures the fraction of the vehicle that has depreciated (between 0 and 1). A constant rate of depreciation means that *Wear* is a nonlinear function of age, but then the price per unit of *Wear* does not depend on its amount. This constant price is estimated for each vehicle type using hedonic price regressions below. Next, in order to separate this choice of vehicle attribute from the choice of vehicle, we assume that the discrete choice is about a brand-new "concept vehicle." Then the household gets reimbursed by the price of *Wear* for accepting an older car. In other words, in our model, a household makes simultaneous decisions about which concept vehicles, how old, and miles to drive.

As it turns out, results for all continuous demands are broadly similar for the sequential and simultaneous models. For discrete choices, however, our simultaneous model finds substantially larger effects from a change in the gas price per mile, income, or vehicle-specific costs. Signs of some elasticities are reversed. In other words, household-specific heterogeneity does affect discrete choices.

The next section describes a behavioral choice model for one-vehicle households and then extends it to consider two-vehicle bundles. It also presents a new method designed for jointly estimating all discrete and continuous choices. Section II describes data sources and provides summary statistics, while III provides estimation results for both discrete and continuous demands. Section IV compares elasticities, and V concludes.

I. The Model and Estimation

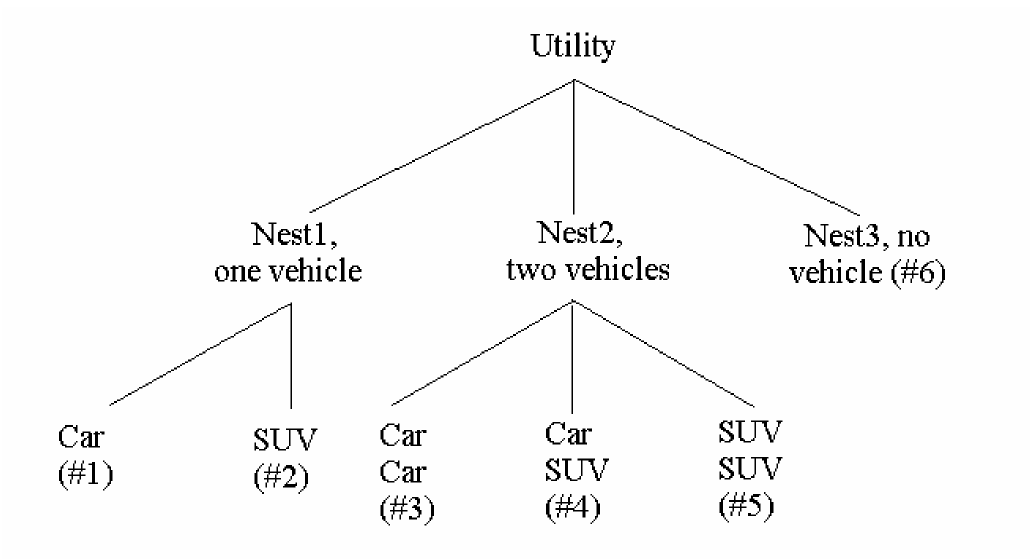
In our model, an agent representing each household faces a discrete choice among a finite number of vehicle bundles. The nesting structure is shown in Figure 1. One

[12] Older vehicles have higher emissions both because older vintages were produced to weaker standards and because pollution control equipment deteriorates with age. Panel data would be required to distinguish these.

[13] Fullerton and West (2000) also simulate effects of incentives in a model of heterogeneous households' continuous choices of car size, car age, and *VMT*, but they use calibrated rather than estimated parameters. That model avoids discrete choices, but it considers only one car per agent. In our model, we estimate discrete choices to consider the household's number of vehicles.

choice is the number of vehicles (0, 1, or 2), and another choice for each vehicle is the type of vehicle (a car or an SUV). We thus have six final bundles, as shown in the figure and listed in Table 1. Other choices important for emissions of each vehicle are the continuous choice about vehicle miles traveled (*VMT*) and vehicle age. To obtain a choice variable with a linear price, we construct "*Wear*" as the fraction of the vehicle used up by depreciation. It is calculated for each car in our sample by assuming 20% depreciation per year, so $Wear = 1 - (1 - 0.2)^{age}$. Thus, a new car has $Wear = 0$.

Figure 1: Nesting Structure for Choice among Vehicle Bundles



Then, since choice of age is considered separately, each discrete vehicle bundle must be defined in a way that is independent of age. For this reason, we define each "concept" vehicle as a bundle of attributes of a brand-new vehicle (car or SUV). The household must pay the price of that brand-new vehicle (the "capital cost"), but then it gets back some money for accepting *Wear* on that vehicle (the "reimbursement" price of *Wear*).

Our demand system now has several distinguishing characteristics. First, it incorporates all of these discrete and continuous choices simultaneously. Second, some unobserved characteristics might affect both kinds of choices. For example, an agent who lives far from work may drive more and thus prefer a larger, more comfortable car. Yet, a more comfortable car may increase the satisfaction of driving and thus induce the driver to drive more. Third, many households have two cars with multiple continuous choices. Consequently, the substitution structure in *VMT* and *Wear* among different vehicles is important in order to understand the effects of policy on driving behavior.

Since the discrete choice in Dubin and McFadden (1984) involves only two alternatives, that paper can use a simple logit model. Our model has six choices, however, and so we require a more general logit structure. We use the nested logit. The next sub-section describes the simple case for households with only one vehicle, and the second subsection considers multi-vehicle households. In the third and fourth sub-sections, we discuss the estimation procedure and elasticity calculations.

A. Our Model of Car Choice and Miles Driven

This description starts with the choices of *VMT* and *Wear*, assuming that a one-car household has already chosen vehicle number-and-type bundle i. Given bundle i, an agent's direct utility is a function of *VMT*, *Wear*, and another consumption good c. That is, $U = U(VMT_i, Wear_i, c_i)$. Given income y, the budget constraint is given by:

$$\frac{p_g}{MPG_i} VMT_i - q_i Wear_i + c_i = y - r_i , \tag{1}$$

where p_g is the price of gasoline (in dollars per gallon), and MPG_i is fuel efficiency (in miles per gallon), so that $p_i \equiv p_g/MPG_i$ is the marginal price per mile in the i^{th} vehicle bundle. The "reimbursement" price of *Wear* for vehicle type i is denoted as q_i. The price of the other consumption good is normalized to be 1. The annualized capital cost of the concept-vehicle bundle is r_i. Thus, gasoline is the only cost per mile, whereas capital cost is a fixed cost of each bundle.[14] The indirect utility for bundle i is a function of household income and prices, denoted as $V(y\text{-}r_i, p_i, q_i)$.

One common way to obtain the indirect utility function is to use parametric demand and then solve a system of partial differential equations using Roy's identity (Hausman, 1981). For comparability with other studies, we want *VMT* demand as a log-linear function of the price per mile p_i, available income $y - r_i$, and a vector of observed socio-demographic variables x. We then add the reimbursement price q_i to that equation to get:

$$\ln(VMT_i) = \alpha_V^i + \alpha_p^i p_i - \alpha_q q_i - \beta(y - r_i) + x'\gamma + \eta , \tag{2}$$

where η represents an agent-specific unobserved factor (see below). Also, we assume

$$r_i = (\delta+\rho)k_i, \tag{3}$$

[14] Time variation in gasoline prices may cause time variation in used vehicle prices. Our use of cross-section data helps avoid this problem.

where k_i is the total capital value of bundle i (depreciated or market value), δ is the annual rate of further depreciation in value, and ρ represents the interest and maintenance cost. When we plug (3) into (2) and integrate, the implied indirect utility is:

$$V_i = \frac{1}{\beta}\exp\left(-\alpha_0^i + \beta y - \beta_1 k_i - x'\gamma - \eta\right) - \frac{1}{\alpha_p^i}\exp(\alpha_p^i p_i - \alpha_q q_i) + \varepsilon_i\,, \tag{4}$$

where $\beta_1 = \beta(\delta+\rho)$.[15] This equation includes an extra additive error ε_i that is bundle-specific. As in the usual discrete choice model, this error term represents the difference between true individual utility at choice i and the calculated utility level.[16] For households who choose the no-vehicle bundle #6, continuous variables such as p_i, q_i, and VMT_i are unobservable. Implicitly, we assume that these households may purchase a bicycle or a fare card for public transportation with a fixed fee, similar to the capital cost k_i. With no cost per mile or of *Wear*, their second exponential term in (4) is 1.0. Their capital cost k_i is unobserved, so $\beta_1 k_i$ and α_0^6 are not separately identifiable. Since we allow for a choice-specific intercept, however, we combine both terms into one constant, α_0^6.

Note that the simple addition of $\alpha_q q_i$ to equation (2) dictates the form of indirect utility in (4). This indirect utility then implies specific forms for both demands:[17]

$$\ln(VMT_i) = \alpha_V^i + \alpha_p^i p_i - \alpha_q q_i - \beta y + \beta_1 k_i + x'\gamma + \eta \tag{5a}$$

$$\ln(Wear_i) = \alpha_W^i + \ln\left(\alpha_q / \alpha_p^i\right) + \alpha_p^i p_i - \alpha_q q_i - \beta y + \beta_1 k_i + x'\gamma + \eta \tag{5b}$$

This specification has pros and cons. One limitation is the use of specific functional forms, but these log-linear forms are comparable to prior literature and allow for two different demand functions (5a,b) that are consistent with a single indirect utility function (4). An advantage of this specification is that it allows the price of *Wear* (q_i) to enter the *VMT* demand, and price of *VMT* (p_i) to enter the *Wear* demand, but a

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[15] Our model provides estimates of β and β_1, and these can be used to calculate $(\delta+\rho)$, but we do not provide separate estimates of δ and ρ. Some of our steps below require an assumption about δ, and we use 20 percent for this purpose. Estimates of the depreciation rate for automobiles range from 33% (Jorgenson, 1996) or 30% (Hulten and Wykoff, 1996) to 15%, the rate implicit in the vehicle depreciation schedule currently used by the Bureau of Economic Analysis. We use 20% because it falls between these bounds.

[16] Also, because of this integration, note that the intercept in (4) may be different from the intercept in (2).

[17] More general demand functions such as translog demand or the almost ideal demand system imply much more complicated indirect utility functions that could not be estimated. Also, note that no-vehicle households have zero marginal prices, so they have constant miles traveled (conditioned on observed socio-demographic variables and total income). Thus, no continuous demand equations are needed for these households.

limitation is that the expression $\alpha^i_p p_i - \alpha_q q_i$ enters both demands the same way.[18] Also, both continuous demands have the same income effect, β. A more general model could not be estimated. Note, however, that we have added generality where it matters most. In particular, the price per mile has a bundle-specific coefficient (α^i_p), to allow for different effects on the demand for miles in each type of vehicle. Thus a gas tax might decrease miles in an SUV more than in a car, in a way that depends on fuel efficiency, and the change in miles of a two-car household can differ from the change in miles of a household with two SUV's (or one car and one SUV).

B. Two-Vehicle Households

So far, the model above considers only one vehicle, but many households have two vehicles and thus two continuous choices of miles and two continuous choices of *Wear*. We have the observed *VMT* and *Wear* for each vehicle, so we can incorporate all four continuous choices.[19] The direct utility for a two-vehicle household choosing bundle i is $U(VMT_{i1}, VMT_{i2}, Wear_{i1}, Wear_{i2}, c_i)$. The budget constraint is given by:

$$\frac{p_g}{MPG_{i1}} VMT_{i1} + \frac{p_g}{MPG_{i2}} VMT_{i2} - q_{i1}(Wear_{i1}) - q_{i2}(Wear_{i2}) + c_i = y - r_i, \tag{6}$$

where q_{ij} are reimbursement prices for *Wear* in the two vehicles of bundle i (j = 1, 2). Also, $p_{ij} \equiv p_g/MPG_{ij}$ is the price per mile using the j^{th} car of bundle i. We consider the indirect utility function as follows:

$$V_i = \frac{1}{\beta}\exp\left(-\alpha_0^i + \beta y - \beta_1 k_i - x'\gamma - \eta\right) - \frac{1}{\alpha_{p1}^i}\exp(\alpha_{p1}^i p_{i1} + \alpha_{p2}^i p_{i2} - \alpha_{q1} q_{i1} - \alpha_{q2} q_{i2}) + \varepsilon_i \tag{7}$$

The indirect utility in (7) is similar to (4) except for two extra terms related to the second vehicle's gasoline price p_{i2} and reimbursement price q_{i2}. By Roy's identity, given that the household has chosen bundle i in (7), the four continuous demands are:

$$\ln(VMT_{i1}) = \alpha_{V1}^i + \alpha_{p1}^i p_{i1} + \alpha_{p2}^i p_{i2} - \alpha_{q1} q_{i1} - \alpha_{q2} q_{i2} - \beta y + \beta_1 k_i + x'\gamma + \eta \tag{8a}$$

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[18] Thus, a change in p_i must have the same effect on *Wear* that it has on miles. We tried other models, including one where indirect utility has separate terms $\exp(\alpha^i_p p_i)$ and $\exp(\alpha_q q_i)$, so that p_i would have no effect on *Wear*, and q_i would have no effect on *VMT*. That model would not converge, and anyway it is restrictive by assuming no cross-price effects. We also tried models with more coefficients, to relax these restrictions, and we tried many starting points, but only the model in (4) and (5) could be estimated simultaneously for discrete and continuous choices (especially for two-vehicle bundles considered below).

[19] Another interesting question is about each household member's choice of miles driven (in either car), but we have no such data. As described below, we have only data on miles driven in each vehicle.

$$\ln(VMT_{i2}) = \alpha^i_{V2} + \ln(\alpha^i_{p2} / \alpha^i_{p1}) + \alpha^i_{p1} p_{i1} + \alpha^i_{p2} p_{i2}$$

$$- \alpha_{q1} q_{i1} - \alpha_{q2} q_{i2} - \beta y + \beta_1 k_i + x'\gamma + \eta \tag{8b}$$

$$\ln(Wear_{i1}) = \alpha^i_{W1} + \ln(\alpha_{q1} / \alpha^i_{p1}) + \alpha^i_{p1} p_{i1} + \alpha^i_{p2} p_{i2}$$

$$- \alpha_{q1} q_{i1} - \alpha_{q2} q_{i2} - \beta y + \beta_1 k_i + x'\gamma + \eta \tag{8c}$$

$$\ln(Wear_{i2}) = \alpha^i_{W2} + \ln(\alpha_{q2} / \alpha^i_{p1}) + \alpha^i_{p1} p_{i1} + \alpha^i_{p2} p_{i2}$$

$$- \alpha_{q1} q_{i1} - \alpha_{q2} q_{i2} - \beta y + \beta_1 k_i + x'\gamma + \eta \tag{8d}$$

These demands generalize those of a one-vehicle household in (5) by including terms for p_{i2} and q_{i2} (and so we refer to (8) for "all" demands). The demand for VMT_{i2} is symmetric to VMT_{i1} in explanatory variables, but it is non-linear in parameters of both p_{i1} and p_{i2}. The demands for $Wear_{ij}$ (j= 1, 2) are similarly defined.

C. A Procedure to Estimate Discrete and Continuous Demands Simultaneously

Note that the same parameters appear in both discrete and continuous choice functions, yet previous literature has estimated these choice models separately. Often the estimates for the same parameters are different not only in magnitude but also in sign. In this sub-section, we propose a procedure for simultaneous estimation of bundle choice, vehicle age, and miles driven. We start with separate discussion of car choice and miles driven, and then how we combine them in a single estimation procedure.

Following McFadden's random utility hypothesis, vehicle bundle i is chosen if and only if: $V_i \geq V_j$ for all $j \neq i$. The unconditional expected share for bundle i then is:

$$S_i = \int \Pr(V_i > V_j, \forall j \neq i \mid \eta) f(\eta) d\eta, \tag{9}$$

where S_i is the share choosing bundle i, and $f(\eta)$ is the probability density function of the agent-specific error η. We are now in a position to describe the importance of η. On the one hand, individual heterogeneity represented by η could directly affect the choice of bundle. On the other hand, observed demands for VMT and $Wear$ are conditional on that choice. Since the choice of vehicle bundle is endogenous, the estimated demands for VMT and $Wear$ could be biased if the influence of η in (9) is ignored. In the model of Dubin and McFadden (1984), the error term η can be cancelled out from the inequality $\{V_i>V_j, \forall j\neq i\}$, which simplifies the calculation of probabilities (that is, the integration over η in equation (9) is not necessary). In such a model, η appears only in the continuous demands, so this individual heterogeneity does not affect the choice of vehicle bundle directly. They can estimate the discrete model with error ε_i for each bundle, and then, given predicted bundle shares, they estimate the continuous choices with errors η.

Yet, our purpose here is to *retain* individual-specific heterogeneity η and its effect on bundle choice. Thus, the evaluation of probabilities in our model involves integration over all error components (ε, η), where $\varepsilon = (\varepsilon_1, \varepsilon_2, \ldots, \varepsilon_J)$, and where J is the number of possible vehicle bundles. In our model, the ε_i are assumed to be distributed with a generalized extreme value (GEV) distribution, and η follows an unknown distribution with a zero mean across individuals. Conditional on η, we integrate over the GEV distribution to obtain conditional choice probabilities as a general nested logit model:

$$\Pr\left(V_{ni} > V_{lm}, \forall m \neq i, \forall n, l \middle| \eta\right) = \frac{\exp(V_i/\lambda_n)\left(\sum_{j\in B_k}\exp\left(V_j/\lambda_n\right)\right)^{\lambda_n - 1}}{\sum_{l=1}^{K}\left(\sum_{j\in B_l}\exp\left(V_j/\lambda_l\right)\right)^{\lambda_l}} \quad , \tag{10}$$

where n and l represent nests, i is an alternative within nest n, m is an alternative within nest l, K is the total number of nests, and B_l ($l = 1, \ldots, K$) represents a nested subset of alternatives. Our nesting structure is illustrated in Figure 1.

We also integrate over the distribution of η to obtain unconditional probabilities. The literature offers no guidance on the distribution of the η.[20] To reduce the numerical difficulty in estimation, we let η be uniformly distributed in the interval [$-\xi$, ξ]. We search for the ξ that yields a likelihood function with the largest value.[21]

As pointed out by Dubin and McFadden (1984), the random error η does not have a zero mean conditional on each chosen bundle, due to the endogeneity of bundle choice. This can be seen clearly if we rewrite equations (8a-d) into a more convenient form for estimation (using just equation 8a, as an example):

$$\begin{aligned}\ln(VMT_{i1}) = &\sum_j \alpha_{V1}^j d_{ij} + \sum_j \alpha_{p1}^j p_{j1} d_{ij} + \sum_j \alpha_{p2}^j p_{j2} d_{ij} \\ &- \alpha_{q1}\sum_j q_{j1} d_{ij} - \alpha_{q2}\sum_j q_{j2} d_{ij} - \beta y + \beta_1 \sum_j k_j d_{ij} + x'\gamma + \eta\end{aligned} \tag{11a}$$

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[20] Dubin and McFadden (1984) assume η has a particular form of mean and variance, in order to derive an explicit conditional expectation.

[21] This search yields ξ equal to 0.65. Since the estimation of the logit model requires integration over the individual heterogeneity term η, our model is a mixed logit model (McFadden and Train, 2000).

where d_{ij} is a choice indicator variable equal to one when $i = j$, and where equations (11b-d) are analogous. The random error η is correlated with the choice indicators d_{ij}. Dubin and McFadden (1984) suggest sequential estimation to solve this endogeneity problem (a procedure later adopted by Goldberg (1998) and West (2004)). First, the discrete choice model is estimated and the predicted probabilities are calculated. They then suggest three alternative methods that yield consistent estimates of parameters for continuous demands: the instrumental variable method (IV), the reduced form method (RF), and the conditional expectation correction method (CE). They derive the correction terms in terms of probabilities for the CE method based on the assumption of an *i.i.d.* extreme value distribution of ε_i. However, since we assume a GEV distribution of ε_i, these correction terms cannot be used in our model. We want a method that can be used both for sequential estimation and for our simultaneous estimation, in order to compare them, and so we employ the RF method. Taking expectation of (11a) over η, we have:

$$\begin{aligned}\ln(VMT_{n1}) = &\sum_j \alpha_{V1}^j S_{nj} + \sum_j \alpha_{p1}^j p_{j1} S_{nj} + \sum_j \alpha_{p2}^j p_{j2} S_{nj} \\ &- \alpha_{q1} \sum_j q_{j1} S_{nj} - \alpha_{q2} \sum_j q_{j2} S_{nj} - \beta y + \beta_1 \sum_j k_j S_{nj} + x'\gamma + u_{n1}\end{aligned}, \tag{12a}$$

where S_{nj} is the probability of individual n choosing vehicle bundle j from (9), u_{n1} is an additional error to represent the difference between observed VMT and predicted VMT, and where (12b-d) are analogous (not shown here). The sequential RF method applies least squares to (12a-d), except that the shares S_{nj} are replaced by estimated shares $\hat{S}_{nj}$ from the discrete choice model. In contrast, we estimate (9) and (12a-d) simultaneously.

Since the same parameters appear in both discrete and continuous choice functions, we propose a joint estimation method to capture this simultaneity. In particular, we obtain a set of parameters that maximize the following objective function:

$$\begin{aligned}&F(\Theta | y, p_1, p_2, q_1, q_2, k, x) = -\sum_n (\ln(VMT_1) - f_1)^2 - \sum_n (\ln(VMT_2) - f_2)^2 \\ &- \sum_n (\ln(Wear_1) - g_1)^2 - \sum_n (\ln(Wear_2) - g_2)^2 + \sum_n \ln L\end{aligned}, \tag{13}$$

where f_1, f_2, g_1, and g_2 represent the right hand sides (without the random error u_{n1}) of the four equations (12a-d), $\ln L$ is the log likelihood function of the nested logit, and Θ represents the set of parameters to be estimated by maximizing equation (13).

As is consistent with Dubin and McFadden (1984) and other papers in this literature, the maintained hypotheses are that the utility functional form is correct and that consumers maximize it. Under these hypotheses, our procedure produces consistent estimates of parameters. The reasoning is as follows: if the components of (13) were maximized separately, and if some single set of parameters were the solution to all those separate maximizations, then this set of parameters would also maximize the combined objective function. To compare the results, we estimate our model by both the sequential method and the simultaneous estimation method.

D. Elasticities

Once we obtain the parameter estimates, we are ready to calculate elasticities. To see the marginal effects of prices on indirect utility, and therefore on bundle choice, we use equation (7) to obtain explicit formulas for those derivatives. First, define $\exp(\cdot) \equiv \exp(\alpha^i_{p1} p_{i1} + \alpha^i_{p2} p_{i2} - \alpha_{q1} q_{i1} - \alpha_{q2} q_{i2})$. Then:

$$\frac{\partial V_i}{\partial p_{i1}} = -\exp(\cdot) \ , \qquad \frac{\partial V_i}{\partial p_{i2}} = -\frac{\alpha^i_{p2}}{\alpha^i_{p1}} \exp(\cdot) \tag{14a}$$

$$\frac{\partial V_i}{\partial q_{i1}} = \frac{\alpha_{q1}}{\alpha^i_{p1}} \exp(\cdot) \, , \qquad \frac{\partial V_i}{\partial q_{i2}} = \frac{\alpha_{q2}}{\alpha^i_{p1}} \exp(\cdot) \tag{14b}$$

and the marginal effects of income or capital cost on utility take similar forms:

$$\frac{\partial V_i}{\partial y} = \exp(-\alpha^i_0 + \beta y - \beta_1 k_i - x'\gamma - \eta) \tag{15a}$$

$$\frac{\partial V_i}{\partial k_i} = -\frac{\beta_1}{\beta} \exp(-\alpha^i_0 + \beta y - \beta_1 k_i - x'\gamma - \eta) \tag{15b}$$

Then we derive the elasticity of choice i with respect to a change in variable z_j (where z_j may be any of the price variables, income y, or capital cost k_j):

$$\frac{\partial S_i}{\partial z_j} \cdot \frac{z_j}{S_i} = \frac{\partial S_i}{\partial V_j} \cdot \frac{\partial V_j}{\partial z_j} \cdot \frac{z_j}{S_i} \tag{16}$$

Since these formulas involve the unconditional probability of vehicle bundle i, calculating each bundle elasticity requires integration over η. In contrast, calculations of *VMT* elasticites do not involve integration over η. For bundle i (i = 1, …, 5), the own- and cross-price elasticities of *VMT* demand are calculated by:

$$e^i_{V1p1} = \frac{\partial \ln(VMT_{i1})}{\partial \ln p_{i1}} = \alpha^i_{p1} p_{i1} = e^i_{V2p1} \ , \qquad e^i_{V2p2} = \frac{\partial \ln(VMT_{i2})}{\partial \ln p_{i2}} = \alpha^i_{p2} p_{i2} = e^i_{V1p2} \tag{17}$$

The elasticities of demand for *Wear* with respect to its price have a similar form:

$$e^i_{W1q1} = \frac{\partial \ln(Wear_{i1})}{\partial \ln q_{i1}} = -\alpha_{q1} q_{i1} = e^i_{W2q1} \ , \quad e^i_{W2q2} = \frac{\partial \ln(Wear_{i2})}{\partial \ln q_{i2}} = -\alpha_{q2} q_{i2} = e^i_{W1q2} \tag{18}$$

We can also calculate the income elasticity, given by:

$$e^i_{Vy} = \frac{\partial \ln(VMT_{i1})}{\partial \ln y} = \frac{\partial \ln(VMT_{i2})}{\partial \ln y} = -\beta y \ , \tag{19}$$

and the total capital cost elasticity, given by:

$$e^i_{Vk} = \frac{\partial \ln(VMT_{i1})}{\partial \ln k_i} = \frac{\partial \ln(VMT_{i2})}{\partial \ln k_i} = \beta_1 k_i \, . \tag{20}$$

In equations (16) – (20), elasticities are typically evaluated at each bundle's mean values of y and k, the bundle average of gas prices per mile (p_1 and p_2) and the bundle average of reimbursement prices (q_1 and q_2).

II. Data and Summary Statistics

In order to analyze household choice of vehicles, miles driven, and vehicle *Wear*, we need micro-data on household characteristics, household income or expenditures, and detailed information about household-owned vehicles such as the number of vehicles, miles driven in each, and vehicle characteristics (including miles per gallon, MPG, and emissions per mile, EPM). No single data set contains all such information.

The Consumer Expenditure Survey (CEX) provides data on household income, characteristics, and household-owned vehicles.[22] For each household, we aggregate expenditures over four quarters, taking demographic data and detailed vehicle information from their last quarter in the survey. We use the CEX from 1996 to 2000, supplemented with the corresponding OVB file (Owned Vehicles Part B Detailed questions). This OVB file includes data on each vehicle type, make, year, number of cylinders, purchase expenses and financing, time since purchase, mileage, gasoline expenditure, and other information. We keep only households that satisfy several criteria. First, expenditures

[22] The CEX data are collected by the Bureau of Labor Statistics of the U.S. Department of Labor through quarterly interviews of selected households throughout the U.S. Each household is interviewed over five consecutive quarters. Each quarter, 20% of households complete their last interview and are replaced by new households. For CEX data, see http://elsa.berkeley.edu or http://www.icpsr.umich.edu/.

must be reported consecutively for four quarters in the CEX of 1996-2000. Second, the household must possess the same number of vehicles during these four quarters. Third, we remove households that own more than two vehicles.[23] We also remove households that have vehicles other than automobiles or SUV's (defined to include light trucks or vans). Finally, we are left with 9027 households, of which 2077 own no vehicles, 4211 own one vehicle, and 2739 own two vehicles. We use yearly total expenditure as a proxy for yearly income of each household. Table 2 defines all the variables used in estimations.

Summary statistics are shown in Table 3 for major household characteristics by vehicle bundle. This table shows significant variations in household characteristics across the number of vehicles and bundles. For example, larger households especially with more kids have more vehicles and prefer SUVs. Wealthier households (as measured by total yearly expenditures) possess more vehicles. Households with more workers or income earners have more vehicles. Households with male heads are inclined to have SUVs.

Next, fuel price data are obtained from the ACCRA cost-of-living index for 1996-2000. This index compiles quarterly data for approximately 300 cities in the United States. It also lists average gasoline price for each city for each survey quarter. Since the CEX reports region and state of residence instead of city for each household, we average the city gas prices to obtain a state price for each calendar quarter. For those states reported in the CEX, but not reported in the ACCRA index, we use the average region price as a substitute. Then we assign a gas price to each CEX household based on the state of residence, CEX quarter, and year.

Some of the variables in our model require calculations or additional sources of data. We now describe these extra calculations.

(1) Wear: The vehicle's age is derived by taking the year of the survey minus the year the vehicle was made. We then assume 20% annual depreciation, and calculate *Wear* as the percentage of the vehicle's value that has wasted away (given all the vehicle characteristics unchanged except vehicle age). *Wear* ranges from zero for a new car, to $Wear = 1$ for a very old car. Specifically, $Wear = 1 - (1 - 0.2)^{age}$.

(2) Capital value of the vehicle: The vehicle's year of purchase and reported purchase price (*pp*) are available in the OVB file, but we want an estimate of current

[23] In the CEX of 1996-2000, 18.4% of households own more than two vehicles. Some of these households may have a vehicle for business, whereas our model of household choice assumes utility maximization.

market value (*cmv*). We calculate the number of "years since purchase" (*ysp*), and we subtract depreciation for each year, again using 20% as the annual rate of depreciation. The formula is $cmv = pp \times (1 - 0.2)^{ysp}$. We then estimate a simple hedonic price regression:

$$cmv = a_0 + a_1 cyl + a_2 im + b_0(1 - Wear) + b_1(Wear \times cyl) + b_2(Wear \times im) \tag{21}$$

where a_0 through a_2, and b_0 through b_2 are parameters. The variable *cyl* denotes the number of cylinders, while *im* is a dummy variable indicating if the vehicle is imported.[24] *Wear* is included in the regression to capture the effects of vehicle age on market value. Using a sub-sample of the CEX that has all necessary variables, we run separate regressions for cars and SUV's and report the results in Table 4. Then, for the value of each brand new "concept" vehicle (with $Wear = 0$), we use:

$$\hat{k} = \hat{a}_0 + \hat{a}_1 cyl + \hat{a}_2 im + \hat{b}_0 \quad . \tag{22}$$

where $\hat{a}_0$ through $\hat{a}_2$ and $\hat{b}_0$ are estimates of parameters in (21).

(3) The price of Wear: First, we calculate the extra amount paid for a car with no wear on it ($Wear = 0$) compared to a very old car with the same characteristics ($Wear = 1$). From (21), that difference is $\left(\hat{b}_0 - \hat{b}_1 cyl - \hat{b}_2 im\right)$. Then, q is the annual reimbursement price of *Wear*, that is, the amount saved during a year by an owner who accepts one whole unit of *Wear* (an old car instead of a new car). Since a very old car does not depreciate any further, the amount saved is the depreciation during the year from holding a new car. Again assuming 20% depreciation, we have: $q = 0.2\left(\hat{b}_0 - \hat{b}_1 cyl - \hat{b}_2 im\right)$.

(4) Fuel Efficiency: The EPA reports miles per gallon (MPG) of new vehicles, but we need it for vehicles of all ages. The CEX does not contain this information, so we estimate MPG using data of the California Air Resources Board (CARB, 1997 and 2000).[25] Their first sub-sample is "series 13", from November 1995 to March 1997, in which the CARB tested a total of 345 passenger cars, light-duty trucks, and medium-duty vans. The second sub-sample is "series 14", from November 1997 to August 1999, which includes

[24] The CEX does not include the vehicle's nation of origin, so we create the *im* dummy using information on manufacturer and model. We also tried other vehicle characteristics in the regression, such as indicators for automatic transmission, power steering, and air conditioning, but the estimates are not significant. Inclusion of these variables does not raise adjusted R^2 and can result in negative predictions of *cmv*.

[25] For MPG of new cars, http://www.fueleconomy.gov/feg/index.htm is a website of the US Environmental Protection Agency (EPA) and the Department of Energy. The EPA also provides the historical fuel economy of new vehicles at http://www.epa.gov/otaq/mpg.htm or at http://www.epa.gov/otaq/tcldata.htm.

332 vehicles (but which reports only 327 vehicles). In total, we use 672 vehicles. We regress MPG against vehicle characteristics in the CARB and then use those estimated coefficients to predict MPG for each vehicle in the CEX. The estimation results are shown in Table 5, where a 4-cylinder SUV is the omitted category. This table shows that fuel efficiency decreases with vehicle age and with engine size, both for cars and for SUV's. Given the same vehicle age and engine size, MPG is higher for cars than for SUV's.

(5) Emissions per mile (EPM): For the same sample of 672 used vehicles, the CARB tests for several pollutants. Following Fullerton and West (2000), we weight each pollutant by estimates of its damages, with the highest weight on nitrous oxides (NO_X, 0.495), followed by hydrocarbons (HC, 0.405), and carbon monoxide (CO, 0.10). Results appear in Table 5. Cars pollute less than SUV's because they were produced under stricter standards. Older vehicles pollute more, both because newer vintages faced stricter standards and because pollution control equipment deteriorates over time.[26]

(6) Vehicle Miles Traveled (VMT): The OVB file provides cumulative miles on each vehicle, but we need yearly miles driven. We had planned to match households across quarters, take the latest odometer reading minus the earliest one, divide by the number of quarters between readings, and multiply by four. Unfortunately, however, some later odometer readings are less than the earlier ones, and many readings are missing. Therefore, we propose a different procedure to get *VMT*. For a one-car household, we take observed annual expenditure on gasoline, divide by the price per gallon to get number of gallons, and then multiply by MPG to get miles. For a two-vehicle household, we only know the total gasoline expenditure, so we need to allocate it between the two vehicles. Only for this allocation do we use the difference in odometer readings between quarters.[27]

(7) Vehicle bundles: As listed in Table 1, vehicle choices are classified into six categories according to the number and type of vehicles. For bundle 4, with one car and one SUV, the car is always identified as the first vehicle. For bundles 3 and 5, the first vehicle is identified as the one with higher yearly *VMT*. If two vehicles have the same

[26] For vehicles in our sample, the calculated *EPM* is 1.89 grams/mile for the average car and 3.56 for the average SUV. It also increases to 6.94 grams/mile for a very old vehicle (with *Wear* =1).

[27] If the difference in odometer readings is positive for both vehicles, then we divide it by MPG to obtain an estimate of each vehicle's gas consumption. Each gasoline amount divided by their sum gives *shares*, used to allocate the observed total gas consumption. Each vehicle's gallons divided by MPG yields *VMT*. If the difference in odometer readings is positive only for one vehicle, we use this figure as VMT_1 and calculate gasoline used in this vehicle. Then total gasoline minus gas used in this vehicle is residual gas, allocated to the other vehicle. Dividing this residual gas by MPG yields VMT_2. If the difference in odometer readings is positive for neither vehicle, then we do imputations based on households with similar characteristics.

yearly *VMT*, the identification is random. If *VMT* is missing, then the vehicle with an earlier purchase year is taken as the first vehicle. If the purchase year and miles-driven are both missing, the identification is random.

III. Estimation Results

The model described in Section I is estimated by both the sequential and the simultaneous estimation methods. The mean values of key variables are reported by bundle in Table 6. We average the values within each bundle for each bundle-specific variable except gas price per mile. Gas price per mile is calculated by dividing gas price per gallon by a bundle-specific MPG listed in Table 1. Thus, gas prices per mile vary both within and between bundles. The presence of collinearity between the fixed effects α_0^i (i = 1, …, 6) and the bundle-specific variables such as k_i (i = 1, …, 5) forces us to normalize the fixed effect of bundle one (α_0^1) to zero. To facilitate the estimation, we also normalize y in units of 10,000 dollars, k_i in units of 1,000, and q_1 and q_2 in units of 100 dollars. Accordingly, we multiply $Wear_1$ and $Wear_2$ by 100 to keep the total amount of reimbursement unchanged in the budget constraint.

Notice that bundle 3 and bundle 5 each contains two vehicles of the same type, while bundle 4 consists of one car and one SUV. When the retail gas price increases, all gas prices per mile are affected in bundle-specific ways because MPG depends both on vehicle age and type (car or SUV). As revealed by Table 1, MPG is more type-specific than bundle-specific. Thus, we expect that the gas price parameters of car bundles 1 and 3 are quite close to one another, as are those of SUV bundles 2 and 5. For a household with one car and one SUV (bundle 4), however, we wish to allow more substitution. In our estimation, we assign one parameter α_{C1} to the gas price of the only car in bundle 1 and first car in bundle 3 (and α_{C2} to the second car). We assign one parameter α_{S1} to the only SUV in bundle 2 and first SUV of bundle 5 (and α_{S2} to the second SUV). Then we assign two gas price parameters to bundle 4: $\alpha_{p1}^4 (= \alpha_{CAR}^4)$ for the car and $\alpha_{p2}^4 (= \alpha_{SUV}^4)$ for the SUV. Results from the sequential estimation are discussed first.

We follow the procedure suggested by Dubin and Mcfadden (1984), but at the first stage we estimate a nested logit structure instead of a multinomial logit model. The traditional ML method is employed. The RF method is adopted at the second stage because the correction terms derived by Dubin and Mcfadden are inappropriate for the

GEV error structure. In the second stage we estimate four continuous demand equations jointly (only two equations for the one-vehicle bundles), using an objective function similar to equation (13) except that the last term is removed. We constrain parameters to be constant across bundles except those for gas prices and constant terms. The estimation results are reported in the first two columns of Table 7, under "sequential estimation".

For the discrete choice model in the first column of Table 7, the estimates of α_{C1} and α_{S1} are significant at the 1% level, while those of α_{C2} and α_{S2} are not statistically significant. The estimates of $\alpha^4_{p1} (= \alpha^4_{CAR})$ and $\alpha^4_{p2} (= \alpha^4_{SUV})$ are both significant at the 0.01 level. All of them are negative as expected. The *Wear* coefficients α_{q1} and α_{q2} are also different from zero at the 0.01 level. The parameter λ_n $(n = 1,2)$ measures the degree of independence of the errors of alternatives in nest n. In our model, the estimates of λ_1 and λ_2 are 0.814 and 0.066, respectively, both significant at the 0.01 level.[28]

Since all the estimates of α_{p1} and α_{p2} are negative, equations (14) indicate that the marginal effects of gas prices per mile are negative. As consistent with expectation, an increase in gas price reduces household utility. Since the coefficient on the reimbursement price q_1 is negative, the marginal effect on utility is positive as expected. A higher reimbursement price means more money back to the household for accepting a given vehicle age or level of *Wear*. However, the coefficient on q_2 has unexpected sign. Since estimates of β and β_1 are both negative and significant, equations (15) indicate that the marginal effect of capital cost is negative while that of income is positive.

We then use those discrete choices from the first column to estimate the continuous demands shown in the second column. A glance down the second column indicates that most of estimated coefficients are quite different from the corresponding estimates in the first column. Yet the parameters in the second column are the same parameters as in the first column, even from the same model, as the continuous demands are supposed to be consistent with a particular indirect utility function. For example, the estimated coefficient on income is −1.408 in the first column and +1.134 in the second column. Both have small errors, and so they are significantly different from each other, even though they are the

[28] If $\lambda_n \; \forall n$ are within the range of zero to one, then "the model is consistent with utility maximization for all possible values of the explanatory variables" (Train, 2003, p.85). Since our λ are significantly less than one, the errors within each nest are correlated, evidence in favor of nesting rather than MNL.

same parameter of the same model. Many price coefficients also differ significantly in magnitude (and the two estimates of α_{q2} differ in sign).

Next, the model is estimated by the simultaneous estimation procedure proposed in Section I.C. The point of this procedure is to capture household-specific heterogeneity in both discrete and continuous choices. The two types of choices are connected by the same parameters and the same random error term η appearing in both.[29] In contrast, in the sequential procedure, the bundle choice affects continuous demands (and not vice versa). The simultaneous estimates are reported in the last column of Table 7.

All ten estimates of coefficients on key variables have the expected signs, and all but two are significantly different from zero. Yet, for many parameters, the estimate differs from *both* estimates obtained by sequential estimation. For example, the capital cost coefficient (β_1) from the simultaneous model (–0.405) is smaller in magnitude than either that of the logit model (–0.671) or the continuous demand model (–0.456). The estimates of coefficients on demographic variables vary with the estimation method, not only in magnitude but also in sign. For most price variables, however, the estimate from the simultaneous model is *between* the two estimates from sequential estimation, which suggests that the simultaneous model might provide more "reasonable" coefficients. These coefficients cannot really be compared directly, however, and so we turn to elasticities.

IV. Elasticity Comparisons

Bundle choice elasticities are presented in Table 8. The upper panel shows elasticities from the sequentially estimated model, but our discussion will start with the elasticities in the lower panel from the simultaneously estimated model. Each entry in the table is not an elasticity with respect to each price in the model, as it might be difficult to interpret an elasticity such as the change in the probability of holding bundle 3 (two cars) for a change in the price p_1 for gas in the first car only. Instead, we calculate the simultaneous effect on all choices for a change in the price of gasoline. In the lower part of Table 8, the first row shows that a 1% increase in the price of gas would decrease most the probability of holding bundle 4 with a car and an SUV (by 0.793%) while increasing the share holding bundle 3 with two cars (by 0.695%). In other words, these households

[29] The standard deviation for $x'\gamma$ is about 0.086 within a bundle, and for βy is about 0.78 within a bundle, so the finding that η has a range (-0.65,0.65) reflects a significant amount of individual heterogeneity. Therefore, introducing individual heterogeneity is expected to make a difference in parameter estimates.

sell the SUV for a second car instead. This change is driven by the high price of driving an SUV with low fuel efficiency.[30] In contrast, using results from the sequential method in the top panel, the price of gas has little effect on any bundle share.

Given vehicle age, a higher reimbursement price q for *Wear* of a particular bundle means more money back to the household and thus higher probability of choosing that bundle. Again, however, it is difficult to interpret a change in the price q_1 for the first car with no change in q_2 for the household's second car. Instead, we show effects of a change in q for all vehicles (or for all cars only, or all SUV's only). Rather than raising q, policymakers may want to reduce q by taxing old vehicles or by subsidizing the purchase of a new vehicle, in order to reduce emissions. Table 5 above shows that emissions per mile (EPM) are higher for SUV's than for cars, and rise with either vehicle's age.

For the simultaneous model in the lower part of Table 8, the second row shows that a 1% tax on *Wear* (lower q for all vehicles) would decrease the probabilities of holding all bundles except bundle 5 (SUV, SUV). In the next row, a tax on the age only of cars would decrease the reimbursement for wear on cars, q_{car}, and switch households out of cars and into bundle 2 with an SUV and bundle 5 with two SUV's. Conversely, the next row shows that a tax on the age only of SUV's that lowers q_{suv} would induce a switch out of bundles 2 and 5 with just SUV's, and into bundles with cars.[31]

The discrete-choice-only model in the top half of the table shows results for q where effects on SUV bundles are unreasonably large and sometimes the wrong sign. A tax that lowers q_{suv} would encourage the purchase of two SUV's.

Back to the lower panel for the simultaneous model, the choice elasticities with respect to y indicate that households with more income switch from holding no car (bundle 6) to one car (bundle 1), and those with a single SUV (bundle 2) seem to add a car (bundle 4). Additional income reduces the share with two cars (bundle 3). These results are inconsistent with the discrete-choice model, where the only bundle with a positive income elasticity is bundle 2 with one SUV.

[30] This reasoning is confirmed by the choice elasticities with respect to p_1 and p_2 separately. For bundle 4, a 1% higher price per mile in the car reduces the probability of choosing that bundle by 0.37%, while a 1% higher price per mile in the SUV (p_2) reduces the probability of choosing that bundle by 0.81%. Thus, the gas consumption of the SUV has twice as much impact as that of the car.

[31] This tax on age of SUV's might actually cut emissions in two ways: by inducing a switch from SUV's to cars (Table 8), and by inducing a switch from older SUV's to newer SUV's (Table 9 below).

We next look at an increase in capital cost in the lower panel of Table 8. Since this change effectively reduces available income, we see that each capital cost elasticity has the opposite sign as that bundle's income elasticity. With higher capital costs, households seem to shift primarily out of two-vehicle bundles with at least one SUV (4 and 5) into bundles with two cars (bundle 3) or only one SUV (bundle 2). While it does not make sense to increase the capital cost only for the first car of a two-car household, it might make sense to increase the capital cost only of cars relative to SUV's or vice versa (to represent a vehicle-type tax). The next row of Table 8 shows that if the increase in capital cost pertains only to cars, then it decreases the shares of the two bundles that have only cars. If it pertains only to SUV's, however, then it has large effects that decrease the shares of all three bundles with SUV's. Such a policy could clearly reduce emissions (given the EPM in Table 5). The 1% higher cost of an SUV means 13.7% less of bundle 4, which seems too large, but it means that the share falls two percentage points (from 14.5% of all households in Table 6 to 12.5% of all households). The discrete-choice-only model in the top part of Table 8 produces elasticities with smaller magnitudes, except that the bundle 5 elasticity has the wrong sign (higher k_{suv} lead to more households with two SUV's).

The sequential model uses predictions of discrete choices to estimate continuous demands, for which elasticities are shown in the top half of Table 9. These are "short run" elasticities, in the sense that car choices are fixed and only continuous choices like driving distances may change (Goldberg, 1998).[32] Again, we focus primarily on simultaneously estimated elasticities in the bottom panel. In the first row, all elasticities for VMT_1 with respect to gasoline price are negative, as expected, for all bundles. (For this demand, the sequential model produces similar results.) The next row of Table 9 shows the effects of a 1% increase in the reimbursement price, q, on *Wear*. These elasticities are all positive, as expected: households choose older vehicles when they get higher reimbursement for holding an old vehicle. Conversely, a tax on vehicle age that reduces q by 10% would reduce desired *Wear* by about 1.2 to 1.4% (assuming the desired cars were available).[33] The table also shows similar effects of changing q just for cars, or just for SUV's.

Next, consider income and capital cost elasticities. Due to the symmetric specification of demand functions, a 1% change in y or k has the same effect on both

[32] Panel data would be required to distinguish the effects of lags from contemporaneous price changes.

[33] In Table 6, the average *Wear* of 0.75 corresponds to 6.2 years of age, so a 1.2% decrease in *Wear* means a decrease of about one month of age. In the sequential model, the same 10% lower q affects desired age of one-vehicle bundles by one-tenth as much, and desired ages of two-vehicle bundles by three times as much.

VMT and *Wear* (whether for the first vehicle or the second). In the simultaneous model, income elasticities are positive as expected. One percent more income would increase driving distances by about 1% to 1.5% for all bundles. In contrast, the sequential model implies income elasticities that are all negative and large (-2.6 to -4.0). The capital cost elasticities are negative as expected, for both models.

The specific form for utility in equation (4) means a specific form for demands in equations (5), where ln(*VMT*) and ln(*Wear*) both depend on $\alpha^i_p p_i - \alpha_q q_i$. In other words, the parameter that determines the important effect of gas price on miles (α^i_p) also necessarily drives the less-important effect of the gas price on choice of *Wear*. Similarly, the own-price effect of q on *Wear* also drives the cross-price effect of q on *VMT*. We note this fact, but we do not mean to emphasize these cross-price elasticities.

Finally, the last column in Table 9 reports the percentage change in total emissions when each variable increases by 1%. In the simultaneous model, for example, a 1% increase in all gasoline prices would reduce total emissions by 0.136%, while a tax on age that reduces q by 1% would reduce total emissions by 0.434%.[34] The largest elasticities are from income and capital cost: 1% higher income raises total emissions as expected, by 4.246% (but in the sequential model would reduce emissions by 11.47%!) A 1% increase in capital cost reduces total emissions by about 8% in either model.

In the simultaneously estimated model, the coefficients are affected by all discrete and continuous choices. The model imposes more constraints on the estimates. Thus, if those constrained estimates are plugged into the likelihood function for either part of the sequential procedure, then the likelihood is not as high as for that *portion* of the sequential procedure. However, the sequentially estimated model yields two sets of estimates for the same parameters. The finding that these estimates are not consistent with each other raises questions about whether the behavioral model is correctly specified.

V. Conclusion

This paper focuses on incentive effects of price changes that might be associated with policies to reduce vehicle emissions. We provide a model of household behavior that incorporates both the discrete choice of vehicle type, with different fuel efficiencies and

[34] These are also short run elasticities, with no change in the number or type of vehicles. Notice that the percentage change in emissions from a change in p is more than twice the change in driving distance, because the higher p also reduces demand for *Wear* (which also reduces emissions). The change in q also affects both *VMT* and *Wear* in the same direction, enlarging the effect on emissions.

emission rates, and continuous demands for miles driven. Because emission rates depend directly on vehicle age, we also model vehicle age as a continuous choice. To model the effect of prices on the choice of vehicle age, we establish a choice of "concept vehicle" that is separate from the choice of "*Wear*". Using hedonic price regressions, we quantify the price of *Wear*. Then, after the discrete choice among concept vehicles, both *VMT* and *Wear* become continuous variables that enter utility.

Yearly household data are obtained from the CEX of 1996 – 2000, supplemented with fuel efficiency estimates from the CARB, and gas prices from the ACCRA cost of living indexes. First, like many others, we follow the sequential procedure suggested by Dubin and McFadden (1984). This procedure generates two different sets of estimates for the same set of parameters, which we argue is inconsistent with maintained hypotheses about the utility function and utility maximization. We then propose and implement a simultaneous method for consistent estimation of both discrete and continuous choices in one step. Results from the simultaneous estimation differ significantly both in signs and magnitude from both sets of estimates obtained by sequential estimation.

We find that a higher price of gasoline would shift households out of the Car-SUV pair and into the bundle with two cars. It also would reduce miles driven. Both of these changes reduce emissions. A tax on vehicle age would induce shifts to newer vehicles with less "*Wear*", and would also shift families out of bundles with an SUV. Both of these changes also reduce emissions. Similarly, a tax on SUV's would shift families into cars and reduce emissions. The size of these shifts is important information for environmental policy. Rather than pin down the exact size of the important parameters, however, this paper points to important problems with existing methods and suggests an alternative approach with more internal consistency.

References

Bento, Antonio, Lawrence Goulder, Emeric Henry, Mark Jacobsen, and Roger von Haefen. "Efficiency and Distributional Impacts of U.S. Policies to Reduce Automobile Pollution", Working Paper, Department of Economics, Stanford University (2005).

Bhat, Chandra. "A Multiple Discrete-Continuous Extreme Value Model: Formulation and Application to Discretionary Time-Use Decisions," Working Paper, Department of Civil Engineering, University of Texas at Austin (2005).

Brownstone, David and Kenneth Train. "Forecasting New Product Penetration with Flexible Substitution Patterns," Journal of Econometrics 89 (1999), 109-29.

Brownstone, D., D.S. Bunch, T.F. Golob, and W. Ren. "A Vehicle Transactions Choice Model for Use in Forecasting Demand for Alternative-Fuel Vehicles," Research in Transportation Economics 4 (1996), 87-129.

California Air Resources Board. Test Report of the Light-Duty Vehicle Surveillance Program, Series 13, Project Number 2S95C1 (September 1997).

California Air Resources Board. Report of the Results of the Vehicle Surveillance Program 14 , Project Number 2S97C1 (March 2000).

Dubin, Jeffrey and Daniel McFadden. "An Econometric Analysis of Residential Electric Appliance Holdings and Consumption," Econometrica 52:2 (March 1984), 345-62.

Eskeland, Gunnar and Shantayanan Devarajan. Taxing Bads by Taxing Goods: Pollution Control with Presumptive Charges, Washington, DC: The World Bank (1996).

Fullerton, Don and Sarah West. "Tax and Subsidy Combinations for the Control of Vehicle Pollution," NBER Working Paper No. 7774, Cambridge, MA (2000).

Fullerton, Don and Sarah West. "Can Taxes on Vehicles and on Gasoline Mimic an Unavailable Tax on Emissions?" Journal of Environmental Economics and Management 43 (January 2002), 135-57.

Goldberg, Pinelopi. "The Regulation of Fuel Economy and the Demand for 'Light Trucks'," Journal of Industrial Economics 46:1 (March 1998), 1-33.

Greene, David L. and Patricia Hu. "The Influence of the Price of Gasoline on Vehicle Use in Multi-vehicle Households," Transportation Research Record 988 (1985), 19-24.

Hanemann, W. Michael. "Discrete/Continuous Models of Consumer Demand," Econometrica 52:3 (May 1984), 54-62.

Harrington, Winston and Virginia McConnell. "Motor Vehicles and the Environment," in H. Folmer and T. Tietenberg, eds., The International Yearbook of Environmental and Resource Economics 2003/2004, Northampton, MA: Edward Elgar (2003).

Hausman, Jerry A. "Exact Consumer's Surplus and Deadweight Loss," American Economic Review 71:4 (September 1981), 662-76.

Heeb, Norbert, Anna-Maria Forss, Christian Saxer, and Patrick Wilhelm. "Methane, Benzene and Alkyl Benzene Cold Start Emission Data of Gasoline-Driven Passenger Cars Representing the Vehicle Technology of the Last Two Decades," Atmospheric Environment 37 (2003), 5185-95.

Hulten, Charles R. and Frank C. Wykoff. "Issues in the Measurement of Economic Depreciation," Economic Inquiry 34 (January 1996): 10-23.

Innes, Robert. "Regulating Automobile Pollution Under Certainty, Competition, and Imperfect Information," Journal of Environmental Economics and Management 31 (September 1996), 219-39.

Jorgenson, Dale W. "Empirical Studies of Depreciation." Economic Inquiry 34 (January 1996), 24-42.

Kohn, Robert E. "An Additive Tax and Subsidy for Controlling Automobile Pollution." Applied Economics Letters 3 (July 1996), 459-62.

Mannering, Fred and Winston, Clifford. "A Dynamic Empirical Analysis of Household Vehicle Ownership and Utilization," Rand Journal of Economics 16:2 (Summer 1985), 215-36.

McFadden, Daniel. "Quantitative Methods for Analyzing Travel Behavior of Individuals: Some Recent Developments," in David Hensher and P. Stopher, eds., Behavioral Travel Modeling, London: Croom Heml (1979), 279-318.

McFadden, Daniel. "Disaggregate Behavioural Travel Demand's RUM Side: A 30-Year Retrospective," in Hensher, ed., Travel Behavior Research: The Leading Edge. London: Pergamon (2001).

McFadden, Daniel and Kenneth Train. "Mixed MNL Models for Discrete Response," Journal of Applied Econometrics 15:5 (2000), 447-70.

Newell, Richard G. and Robert N. Stavins, "Cost Heterogeneity and the Potential Savings from Market-Based Policies," Journal of Regulatory Economics 23:1 (January 2003), 43-59.

Plaut, Pnina. "The Comparison and Ranking of Policies for Abating Mobile-Source Emissions," Transportation Research D 3 (July 1998), 193-205.

Pigou, Arthur C. The Economics of Welfare, London: MacMillan (1920).

Sevigny, Maureen. Taxing Automobile Emissions for Pollution Control. Cheltenham, UK and Northhampton, MA: Edward Elgar Publishing Ltd. (1998).

Sierra Research. "Analysis of the Effectiveness and Cost-Effectiveness of Remote Sensing Devices." Report SR94-05-05, prepared for the U.S. Environmental Protection Agency, Sacramento, CA: Sierra Research (1994).

Train, Kenneth. Qualitative Choice Analysis: Theory, Econometrics, and an Application to Automobile Demand, Cambridge, MA: The MIT Press (1986).

Train, Kenneth. Discrete Choice Methods with Simulation, Cambridge, United Kingdom: Cambridge University Press (2003).

Train, Kenneth, William B. Davis and Mark D. Levine. "Fees and Rebates on New Vehicles: Impacts on Fuel Efficiency, Carbon Dioxide Emissions, and Consumer Surplus," Transportation Research E 33 (1997), 1-13.

U.S. Environmental Protection Agency. Technical Methods for Analyzing Pricing Measures to Reduce Transportation Emissions, Washington, DC (1998).

West, Sarah. "Distributional Effects of Alternative Vehicle Pollution Control Policies," Journal of Public Economics 88:3-4 (March 2004), 735-57.

Table 1. Vehicle Bundle Descriptions and Statistics

Bundle	# of Vehicles	First Vehicle	Second Vehicle	# of Households	MPG of First Vehicle	MPG of Second Vehicle
1	1	Car	--	3469	21.37	--
2	1	SUV	--	742	16.76	--
3	2	Car	Car	1181	21.88	21.55
4	2	Car	SUV	1305	21.51	16.53
5	2	SUV	SUV	253	17.04	16.50
6	0	--	--	2077	--	--

Note: The number of households is from the consumer expenditure survey (CEX), and miles per gallon (MPG) is calculated from CARB data described below.

Table 2. Variable Definitions

Variable	Definition
y	Household's yearly expenditure
k	Total capital cost of a vehicle bundle
p_1	Gas price per mile of the first vehicle
p_2	Gas price per mile of the second vehicle
q_1	Unit price of *Wear* of the first vehicle
q_2	Unit price of *Wear* of the second vehicle
VMT_1	Miles driven in the first vehicle
VMT_2	Miles driven in the second vehicle
$Wear_1$	Continuous variable to measure the wear of the first vehicle
$Wear_2$	Continuous variable to measure the wear of the second vehicle
Famsize	Number of members in a household
Earnr	Number of income earners in a household
Kids	Number of children less than 18 in a household
Drivers	Number of household members 16 years old and over
Metro	A dummy variable: one if the household resides inside a Metropolitan Statistical Area (MSA), and zero otherwise
Pop4	A dummy variable: one if the household lives in an area with a population of more than 4 million, and zero otherwise
Urban	A dummy variable: one if the household lives in an urban area, and zero otherwise.
Age	Age of household head
White	A dummy variable: one if the household head is white, and zero otherwise
Male	A dummy variable: one if the head is male, zero otherwise
Educ	A dummy variable: one if the head has education higher than high school, zero otherwise
Northwest	A dummy variable: one if in the Northwest, zero otherwise
Midwest	A dummy variable: one if in the Midwest, zero otherwise
South	A dummy variable: one if in the South, zero otherwise
West	A dummy variable: one if in the West, zero otherwise

Table 3. Summary of Household Statistics by Vehicle Bundles

Characteristics	Number of Vehicles					
	1		2			0
	1 (Car)	2 (SUV)	3 (C,C)	4 (C,S)	5 (S,S)	6 (none)
# of households	3469	742	1181	1305	253	2077
household size	1.92	2.30	2.65	2.94	3.44	1.98
% with kids	23.87	33.56	33.62	43.98	62.45	26.05
# of kids	0.44	0.73	0.56	0.89	1.42	0.55
# > 15 years old	1.52	1.63	2.13	2.12	2.13	1.48
# of workers	0.85	1.08	1.43	1.49	1.58	0.70
% heads male	40.10	63.07	65.54	71.80	77.47	33.22
age of head	55.24	48.22	51.84	49.45	45.24	55.66
% heads white	82.07	87.60	83.32	89.04	92.89	67.89
% heads educ > high school	52.15	52.29	66.05	57.01	57.31	34.33
% in area with pop.> 4 million	28.37	19.41	30.48	22.68	18.58	38.61
expenditures	22754.	24574.	35472.	33812.	34246.	17795.
total gas cost	648.	920.	1103.	1279.	1398.	--

Table 4. Hedonic Price Regressions

Dependent Variable: *cmv*	Cars		SUVs	
	Coefficient	Standard Error	Coefficient	Standard Error
constant (a_0)	1444.64	1806.08	-1220.52	2702.42
cyl (a_1)	3150.55	288.44	1993.56	411.23
import (a_2)	2371.11	894.32	1417.36	1584.27
1-Wear (b_0)	-2179.03	3272.66	8973.32	4996.71
Wear×cyl (b_1)	-3184.92	546.49	-1459.66	763.85
Wear×import (b_2)	-998.07	1719.28	-658.35	2800.80
R^2	0.49		0.51	
# of obs.	793		510	

Table 5: Estimation of Miles Per Gallon (MPG) and Emissions Per Mile (EPM)

Independent Variable	Dependent Variable			
	MPG		EPM	
	Coefficient	Standard Error	Coefficient	Standard Error
constant	24.021	0.496	-0.597	0.663
cyl6	-4.395	0.483	1.103	0.645
cyl8	-7.948	0.581	3.548	0.777
age	-0.419	0.049	0.285	0.065
age^2	0.006	0.002	0.003	0.002
car	4.262	0.410	-0.589	0.548
cyl6 × car	-1.439	0.560	-0.661	0.749
cyl8 × car	-1.149	0.655	-2.819	0.875
R^2	0.7598		0.4095	
F-value	299.997		65.775	
# of obs.	672		672	

Table 6. Mean Values of Key Variables Involved in Estimation

Variable	Bundle					
	1 (Car)	2 (SUV)	3 (C,C)	4 (C,S)	5 (S,S)	6 (none)
% of households	38.43	8.22	13.08	14.46	2.80	23.01
VMT_1	11799.	12977.	15283.	10513.	16151.	--
VMT_2	--	--	5554.	10771.	5358.	--
price of gas 1 (p_1)	0.058	0.074	0.056	0.057	0.072	--
price of gas 2 (p_2)	--	--	0.057	0.075	0.075	--
vintage1	8.62	8.24	7.63	7.89	6.87	--
vintage2	--	--	9.02	8.50	8.78	--
$Wear_1$	0.76	0.73	0.72	0.73	0.68	--
$Wear_2$	--	--	0.77	0.73	0.75	--
price of $Wear_1$ (q_1)	15572.	18010.	15363.	15686.	18052.	--
price of $Wear_2$ (q_2)	--	--	15301.	18133.	18105.	--
expenditure (y)	22754.	24574.	35472.	33812.	34246.	17795.
capital cost (k)	17224.	20187.	34157.	37684.	40551.	--
capital cost 1	17224.	20187.	17125.	17337.	20232.	--
capital cost 2	--	--	17032.	20348.	20319.	--

Table 7. Estimation Results

Parameters	Sequential Estimation: Nested Logit	Sequential Estimation: Continuous Demands	Simultaneous Estimation
p_{11}, p_{31} (α_{C1})	-0.246**	-0.460**	-0.433**
	(0.025)	(0.070)	(0.073)
p_{32} (α_{C2})	-0.045	-0.238*	-0.045**
	(0.033)	(0.143)	(0.008)
p_{21}, p_{51} (α_{S1})	-0.237**	-0.927**	-0.526**
	(0.028)	(0.054)	(0.105)
p_{52} (α_{S2})	-0.011	-0.453	-0.013
	(0.049)	(0.380)	(0.080)
p_{41} (α^4_{CAR})	-0.240**	-0.374**	-0.399**
	(0.024)	(0.143)	(0.062)
p_{42} (α^4_{SUV})	-0.084**	-1.331	-0.662**
	(0.022)	(1.582)	(0.103)
q_1 (α_{q1})	-0.012**	-0.370E-03	-0.004**
	(0.003)	(0.002)	(0.001)
q_2 (α_{q2})	0.010**	-0.010**	-0.219E-36
	(0.001)	(0.002)	(0.936E-36)
y (β)	-1.408**	1.134**	-0.420**
	(0.086)	(0.134E-03)	(0.001)
k (β_1)	-0.671**	-0.456**	-0.405**
	(0.108)	(0.034)	(0.023)
Choice specific:			
constant 2 (α_0^2)	-1.403**		0.645**
	(0.278)		(0.035)
constant 3 (α_0^3)	4.219**		1.860 **
	(0.516)		(0.031)
constant 4 (α_0^4)	5.057**		2.063**
	(0.650)		(0.051)
constant 5 (α_0^5)	2.401**		2.320**
	(0.685)		(0.062)
constant 6 (α_0^6)	-2.045**		-0.948**
	(0.383)		(0.132)
Demand-Specific:			
constant 1 (α_{V1})		9.578**	0.302**
		(0.179)	(0.087)
constant 2 (α_{V2})		7.361**	0.805**
		(0.187)	(0.088)
constant 3 (α_{W1})		9.346*	2.580**
		(5.007)	(0.298)
constant 4 (α_{W2})		5.147**	5.114**
		(0.176)	(1.259)

(continued on the next page)

Table 7. Estimation Results (cont'd)

Famsize	0.332	0.072**	0.058**
	(0.542)	(0.002)	(0.001)
Earnr	0.270**	0.067**	0.032**
	(0.067)	(0.001)	(0.183E-03)
Kids	0.510	0.081**	-0.031**
	(0.527)	(0.002)	(0.001)
Drivers	0.190	0.060**	-0.041**
	(0.535)	(0.001)	(0.001)
Metro	-0.552**	-0.012**	0.012**
	(0.123)	(0.002)	(0.474E-03)
Pop4	-0.340**	-0.013**	0.012**
	(0.085)	(0.001)	(0.290E-03)
Urban	-0.441**	-0.058**	0.105**
	(0.161)	(0.002)	(0.001)
Age	0.046**	-0.007**	0.004**
	(0.003)	(0.290E-04)	(0.128E-04)
White	0.056	0.136**	0.097**
	(0.091)	(0.001)	(0.386E-03)
Male	0.057	0.109**	0.004**
	(0.085)	(0.001)	(0.240E-03)
Educ	0.020	0.058**	0.036**
	(0.072)	(0.001)	(0.263E-03)
Northwest	0.244	0.042**	0.046**
	(0.179)	(0.001)	(0.386E-03)
Midwest	0.401**	0.064**	0.059**
	(0.173)	(0.001)	(0.380E-03)
South	-0.726**	-0.150**	0.072**
	(0.121)	(0.001)	(0.374E-03)
λ_1	0.814**		0.138**
	(0.053)		(0.006)
λ_2	0.066**		0.103**
	(0.003)		(0.005)
Log Likelihood	-28917.8	-786857	-0.310E+07

* indicates 0.10 significance level, and ** indicates 0.05 significance level.

Table 8. Elasticities of Discrete Choices for each Variable

	Bundle					
Variable	1 (Car)	2 (SUV)	3 (C,C)	4 (C,S)	5 (S,S)	6 (none)
Sequential: [a]						
p	0.015	-0.106	0.006	-0.177E-03	0.034	--
q	-0.207	3.618	-0.116	-0.033	-6.077	--
q_{car}	1.530	-6.318	0.139	0.127	-3.470	--
q_{suv}	-1.737	9.937	-0.255	-0.160	-2.603	--
y	-0.106	0.591	-0.042	-0.006	-0.011	-0.006
k	0.086	-0.427	0.061	0.008	-0.303	--
k_{car}	-0.008	0.127	0.056	-0.944	4.336	--
k_{suv}	0.110	-0.413	0.134	-1.099	4.703	--
Simultaneous: [b]						
p	0.009	-0.073	0.695	-0.793	0.020	--
q	0.025	0.193	0.066	0.283	-0.001	--
q_{car}	0.177	-0.966	0.151	0.352	-0.147	--
q_{suv}	-0.153	1.159	-0.085	-0.069	0.146	--
y	0.341	-1.203	-0.818	0.634	0.010	-0.074
k	-0.321	0.390	1.655	-6.319	-0.377	--
k_{car}	-1.229	7.315	-13.021	7.345	1.263	--
k_{suv}	0.908	-6.925	14.676	-13.665	-1.640	--

[a] Calculation based on estimates in column 1 of Table 7.

[b] Calculation based on estimates in column 3 of Table 7.

Table 9. Short-Run Elasticities of Continuous Demands

	Bundle					Total
Variable	1 (Car)	2(SUV)	3 (C,C)	4 (C,S)	5 (S,S)	Emissions [c]
Sequential: [a]						
p	-0.026	-0.066	-0.038	-0.117	-0.098	-0.211
q	0.012	0.013	0.306	0.360	0.362	0.631
q_{car}	0.012	--	0.306	0.012	--	0.368
q_{suv}	--	0.013	--	0.349	0.362	0.263
y	-2.581	-2.788	-4.024	-3.836	-3.885	-11.472
k	-1.570	-1.840	-3.113	-3.434	-3.695	-8.746
Simultaneous: [b]						
p	-0.024	-0.037	-0.026	-0.070	-0.038	-0.136
q	0.122	0.141	0.120	0.123	0.141	0.434
q_{car}	0.122	--	0.120	0.123	--	0.293
q_{suv}	--	0.141	--	7.933E-36	0.141	0.141
y	0.956	1.032	1.490	1.420	1.438	4.246
k	-1.397	-1.637	-2.770	-3.056	-3.288	-7.783

Each entry is the elasticity of *VMT* or *Wear*, in the first or second vehicle, with respect to each variable.

[a] Calculation based on estimates in column 2 of Table 7.

[b] Calculation based on estimates in column 3 of Table 7.

[c] The last column is the percent change in total emissions, $E = \sum \text{EPM} \times \text{miles}$, adding over all vehicles in all bundles, for a one percent change in each variable.

Market Mechanisms and Incentives: Applications to Environmental Policy

October 17, 2006

Discussant: Ed Coe
<u>Session III: Mobile Sources</u>

<u>Tradable Fuel Economy Credits: Competition and Oligopoly</u>

Given that there is some increased interest in examining options for reducing GHGs from the transportation sector, this study comes at an opportune time. Also, since many groups are examining many different options, it is useful to have a model that can examine a number of different options.

While there are a number of models that exist that can estimate the impacts of changes in CAFÉ standards, the particular strength of this model is its ability to estimate the impacts on particular auto manufacturers. Since this model also examines different platforms, it should be possible to examine impacts if the passenger car CAFÉ standards were set in a fashion similar to the light-duty truck reformed CAFÉ standards, which are based on six platforms. Also, given that some auto manufacturers are exploring the possibility of merging or developing partnerships, this model might be able to assess, to some extent, the impacts of the combined entity.

It is very useful, from a policy perspective, that this model can examine the impacts assuming perfect competition, and oligopolistic approaches. It's useful to note from a policy perspective, that a significant portion of the total savings available is from class averaging within firms – it is important to note this, if one assumes that there might be non-competitive behavior regarding credits.

DOE's NEMS considers a technology to be cost-effective if the technology pays back in three years at a 15% discount rate. It would be interesting to apply those assumptions here and see what kind of impacts they might have on the results.

Other thoughts
Price set by EIA's reference case of $1.51/gallon, and "high B" forecast of $1.84/gallon
Miles driven is fixed for each vehicle class (no rebound?)
No diesel or hybrid technology
No alternative fuel – E85 vehicles?
After the fact FFV credits?

<u>Environmental Marketing of Passenger Vehicles: Strategies and Impacts</u>

It's no secret to anyone that the effectiveness of eco-communications or labeling programs is very difficult to quantify. I think that this study makes a good attempt at attempting to quantify the effectiveness of these types of programs, and at the very least, does show trends.

From a policy and program perspective, the quantification of these types of programs would go a long way in assisting states meeting their State Implementation Plans (SIPs). However, it's not clear to me that such a rigorous model could be developed in the near future, but I'm willing to be convinced otherwise [real reductions, verifiable, enforceable].

I noticed that the methodology is done in a two step process, a person picks the class of vehicle to purchase, then considers information within a class. But, a Maritz study, which is a Car Buyer Market Research firm, recently conducted a study of new car buyers and found that about 1/3 of all new car buyers look across classes. Is it possible to model that behavior?

EPA has developed the Green Vehicle Guide, which is on the Web. Those cars that meet certain air pollution and greenhouse gas emissions criteria get a special designation of SmartWay. It would be interesting to see a pilot program in which a state uses these designations and examine whether the SmartWay label has an effect on consumer choice.

I think that using Auto Dealers as a surrogate for Auto Producers might represent a weakness in the model, since auto dealers cannot develop new product lines, but might be useful from the perspective that they have some control over their inventory.

Would be interesting to see how people would react to today, given a greater awareness or sensitivity to gas prices.

<u>Vehicle Choices: Miles Driven, and Pollution Policies</u>

From a policy perspective, it is important to have a model that can estimate the effectiveness of policies or measures applied to the light-duty mobile sector for reducing criteria pollutant. Again, as in the last study, such a model that can estimate benefits within a certain band of uncertainty, can be useful in the State Implementation Plans context, to the extent that reductions are real, verifiable, and enforceable.

As you mentioned, the particular strength of this model, is its ability to capture the simultaneity of certain decisions and yield a single set of parameters.

You conclude that a higher price for gasoline would tend to shift households out of the Car-SUV pair and into the bundle with two cars. You also conclude that miles driven would be reduced. However, given that the SUV has been replaced by a car, and the cost of driving for that household has been reduced, is it possible that household might be induced to drive a little more?

Why use mpg as a variable instead of gallons-per-100 miles or other fuel consumption metric? mpg vs fuel consumed is non-linear while gallons-per-100 miles vs fuel consumed is a linear relationship.

[i.e. going from 10 to 12 mpg is a much larger fuel savings than going from 30 to 32 mpg, whereas going from 5 to 4 gallons per 100 miles saves the exact same amount of fuel as from 11 to 10 gallons per 100 miles]

Workshop on
Market Mechanisms and Incentives: Applications to Environmental Policy
October 17th and 18th, 2003
Resources for the Future

Mobile Source Session: Discussion

Winston Harrington

The three papers presented in the mobile source session were all high quality papers. Each asked a different question, but all were related. One was concerned with modeling vehicle supply, another with vehicle demand, and the third with whether and how vehicle demand might be shaped by public relations campaigns appealing to altruistic motives.

1. Rubin, Jonathan, Paul Leiby and David Greene, "Tradable Fuel Economy Credits: Competition and Oligopoly"

This is a very nice paper I think, and it generates some interesting results. It's not a welfare analysis and it doesn't compare CAFE to other potential fuel saving policies, but a cost-effectiveness paper focused on CAFE policy design. The authors have built an interesting model of vehicle supply that is both manufacturer and vehicle class-specific. Vehicle Classes are limited to cars and trucks, but that is enough for their purposes. The model allows them to compare the perfectly competitive solution to the Nash-Cournot and Stackelberg oligopoly models. They use the NAS cost assumptions for fuel-saving technologies. The purpose of the paper is to determine the potential cost savings available from various kinds of CAFE credit trading and the extent to which those savings are compromised by imperfect competition. The most important conclusions of the exercise is that (i) one can get most of the benefits of CAFE trading simply by pooling the car and light truck categories, without having trading across manufacturers, and (ii) the cost savings are not much affected by oligopoly.

There were three further aspects of the results that caught my eye. First, in the perfectly competitive case, fully tradable CAFE can achieve cost savings that exceed 100%. That is, fully tradable CAFE can actually reduce costs. The authors observe this, but don't really offer an explanation. Considering that each CAFE technology has positive costs (i.e. no assumptions here of Porteresque efficiency gains from forcing manufacturers to look where they haven't before), this outcome deserves some discussion. One possibility that occurred to me concerns the baseline. The policies they examine are a 30 and 40 percent improvement in CAFE over the current US policy. Of course, the current policy has well-known inefficiencies, so perhaps the costs of more stringent CAFE standards are more than offset by the removal of the inefficiencies of the current CAFE policy.

Second, the authors' estimates of the distributional effects of tradable CAFE are striking and, it seems to me, counter to the conventional wisdom. I don't really understand how US manufacturers, like Ford and GM are not hurt, especially by the pooling of the car and truck categories. Ford's fleet mix is heavily weighted toward truck, so if permit allocations are based on the status-quo fleet, then Ford, with its vehicle fleet heavily weighted toward trucks, would seem to be at a disadvantage. It would be useful for the authors to provide a little intuition of how this could be

Third, the paper makes the point that if the cost of the technology is low, then there is little value to a marketable permit system, because the constraint is barely binding. If the cost is high, then there is little value to a permit market because no manufacturers will have "surplus" permits and there are few gains from trade. This conclusion, I think, is driven by the NAS cost estimates, which do not vary much across categories. Without cost heterogeneity, it is of course true that gains from trade are minimal. But I still think they are selling markets short, because without a mechanism you won't know what the costs are. One of the unsung advantages of markets is that they are effective devices for cost revelation.

2. Ye Feng, Don Fullerton and Li Gan, Vehicle Choices, Miles Driven and Pollution Policies."

This paper tackles a really important methodological problem involving discrete-continuous models of vehicles and use. These models were pioneered by Dubin and McFadden in the study of household appliance demand, and have been used by many authors to study the demand for motor vehicles. These models posit a utility function that yields a demand function for vehicles, and conditional on vehicles owned, a demand function for VMT. These two demand functions have many parameters in common, but in empirical work it has been the usual practice to estimate them not as a system but sequentially, a procedure that provides two distinct estimates of parameters that should be equal. This can be okay if you are simply trying to predict VMT at the household level, and these models do a pretty good job of that. But for other tasks, having what amounts to an *ad hoc* procedure can lead to problems. For example, if you are trying to estimate welfare, these models can lead to nonsensical results.

The reason researchers have not estimated a system of equations is that it has proved to be very difficult to do. Households have a huge number of possible choices for vehicle ownership combinations, and this variety presents real difficulties in estimation. Feng et al. make some innovations and simplications that make the estimation manageable. First, they classify vehicles into only two types: cars and trucks. In addition, Second, they make age a continuous variable, which is distinct from the usual practice of having distinct variables for each vintage. In effect age is turned into a variable that measures the value of the vehicle stock. Third, they limit themselves to only households containing two vehicle or fewer.

With these simplifications they are able to estimate a simultaneous system of equations, and they nicely contrast these results to the results of a sequential model in a table. They show first, how different the two estimates of the same parameter can be in the sequential model, and second, how the simultaneous model results are different from either.

Of course, with the simplifications of the specification there will be costs. The aggregation to two vehicles ignores the role of particular vehicle characteristics in explaining consumer buying behavior, except insofar as they are captured in the car/truck difference. But cars (or trucks) differ greatly in acceleration, number of passengers, towing capacity, interior volume and other features. There is risk here of omitted variable bias. In two-vehicle households the difficulty becomes even more complex, since households looking for a particular feature may only require it in one of their vehicles. The authors defend this assumption by observing that these characteristics do not affect emissions, which is true as far as emissions of conventional pollutants are concerned, but not green house gases.

In addition, the restriction to households with two vehicles or less omits 18% of US households and 33% of all vehicles, which could account for a large share of VMT. In response to this comment at the workshop, it was claimed that the model with three households is just too complex to estimate. It was unclear to me whether this was due to a lack of computing power or something else. In addition, Don speculated that the VMT in the third (or greater) vehicle in the household would be much less than the two primary vehicles, but in fact the data from the 2000 Nationwide Household Travel Survey suggest that the falloff in mileage for the third car is surprisingly small. Perhaps this shouldn't be too surprising, since most of the households owning more than two vehicles also have more than two licensed drivers. What the data suggest is that households respond to the low marginal cost of vehicle operation, and once a vehicle is in the household, it is driven.

If I were to make any suggestions for the authors it would be to revisit the two-vehicle limitation and if possible, extend to allow for three vehicle households. Beyond this, one interesting comparison would be for the authors to estimate the welfare effect of vehicle fuel price, and compare to the welfare change estimated from the sequential model. If their experience is like ours, they will find that the welfare estimates made using the coefficients from the discrete part of the sequential model will be nonsensical.

3. Mario Teisl, Jonathan Rubin, and Caroline Noblet, Do Eco-Communication Strategies Reduce Energy Use and Emissions from Light Duty Vehicles?

This is a very well-conceived project, an experiment to estimate the effectiveness of providing consumers with information about the emission characteristics of new vehicles in pro-bono radio spots. The campaign itself consisted of two parts: a series of radio spots and other PR designed to raise consciousness. One of its striking features is the cooperative venture combining the efforts of state government, automobile dealerships, and environmentalists. As far as I am aware, you rarely see this kind of cooperation in an experiment. Usually the parties want something—PR, action, etc.—that makes it difficult to adhere to a proper experimental design.

The design here is classic. You have a localized treatment area and a control area consisting of the rest of the state. Two surveys conducted before and after a campaign to encourage purchase of environmentally benign vehicles allow the researchers to isolate the effects of the treatment from other influences on vehicle purchase decisions.

A few comments on the paper and the results, as opposed to the experimental design.

1. The paper itself shows signs of being an early draft, and I'm sure with more editing it will improve substantially. For example, the authors don't tell usmuch at all about the statistical approach. At the workshop Jonathan indicated that ordered logit was the statistical model used to analyze the attitudinal questions, but "logit" appears nowhere in the paper. There was also nothing in this draft about the bottom line—the effect on vehicle purchase decisions, and apparently there won't be. In his presentation I believe Jonathan said those results are in a separate paper. To me, it's a little disappointing to separate the results like that, and it belies the title of this paper.

2. The finding that the car dealerships were did not participate in the campaign in spite of the support given to it by their own trade association was surprising but not unprecedented. Karen Palmer has told me of other cases involving battery recycling where the efforts of the national trade association were ignored by the local members. It is a bit depressing; if contacting their trade association doesn't work, then how would it be possible to engage the dealerships?

3. Another outcome of interest was the change in the attitude variables as a result of the campaign. In particular, the variables CONC and AQUAL measured the respondents level of concern and his assessment of current air quality, respectively. Not surprisingly, the level of concern about air quality increased. But I *was* a little surprised that the campaign adversely affected respondents' assessment of current air quality. Large reductions in concentrations of fine particulates and ozone in the last 15 years or so have been one of the signal accomplishments of environmental regulation in the US. Now the question asked was whether air quality was good or bad, which is a bit different from whether it has improved or not. Nonetheless it seems to me that respondents are not getting the full picture of air quality in Maine. Perhaps it is too much to ask that respondents get a more nuanced picture of air quality in a survey such as this.

Market Mechanisms and Incentives: Applications to Environmental Policy

A Workshop Sponsored by the U.S. Environmental Protection Agency's National Center for Environmental Economics (NCEE) and National Center for Environmental Research (NCER)

Resources for the Future
1616 P Street, NW, Washington, DC 20036
October 17-18, 2006

Disclaimer

Market Mechanisms and Incentives: Applications to Environmental Policy

Resources for the Future
1616 P Street, NW, Washington, DC 20036
(202) 328-5000

October 17th – 18th, 2006

October 17, 2006: Market Mechanisms in Environmental Policy

8:00 a.m. – 8:45 a.m. **Registration**

8:45 a.m. – 11:45 a.m. **Session I: Brownfields and Land Issues**
Session Moderator: **Robin Jenkins**, EPA, National Center for Environmental Economics

8:45 a.m. – 9:00 a.m. Introductory Remarks: **Sven-Erik Kaiser**, EPA, Office of Brownfields Cleanup and Redevelopment

9:00 a.m. – 9:30 a.m. Environmental Liability and Redevelopment of Old Industrial Land
Hilary Sigman, Rutgers University

9:30 a.m. – 10:00 a.m. Incentives for Brownfield Redevelopment: Model and Simulation
Peter Schwarz and **Alex Hanning**, University of North Carolina at Charlotte

10:00 a.m. – 10:15 a.m. **Break**

10:15 a.m. – 10:45 a.m. Brownfield Redevelopment Under the Threat of Bankruptcy
Joel Corona, EPA, Office of Water, and **Kathleen Segerson**, University of Connecticut

10:45 a.m. – 11:00 a.m. Discussant: **David Simpson**, EPA, National Center for Environmental Economics

11:00 a.m. – 11:15 a.m. Discussant: **Anna Alberini**, University of Maryland

11:15 a.m. – 11:45 a.m. Questions and Discussion

11:45 a.m. – 12:45 p.m. **Lunch**

12:45 p.m. – 2:45 p.m. **Session II: New Designs for Incentive-Based Mechanisms for Controlling Air Pollution**
Session Moderator: **Will Wheeler**, EPA, National Center for Economic Research

12:45 p.m. – 1:15 p.m. Dynamic Adjustment to Incentive-Based Environmental Policy To Improve Efficiency and Performance
Dallas Burtraw, Danny Kahn, and Karen Palmer, Resources for the Future

	1:15 p.m. – 1:45 p.m.	Output-Based Allocation of Emissions Permits for Mitigating Tax and Trade Interactions **Carolyn Fischer**, Resources for the Future
	1:45 p.m. – 2:00 p.m.	Discussant: **Ann Wolverton**, EPA, National Center for Environmental Economics
	2:00 p.m. – 2:15 p.m.	Discussant: **Arik Levinson**, Georgetown University
	2:15 p.m. – 2:45 p.m.	Questions and Discussion
2:45 p.m. – 3:00 p.m.	**Break**	
3:00 p.m. – 5:30 p.m.	**Session III: Mobile Sources** Session Moderator: **Elizabeth Kopits**, EPA, National Center for Environmental Economics	
	3:00 p.m. – 3:30 p.m.	Tradable Fuel Economy Credits: Competition and Oligopoly **Jonathan Rubin**, University of Maine; **Paul Leiby**, Environmental Sciences Division, Oak Ridge National Laboratory; and **David Greene**, Oak Ridge National Laboratory
	3:30 p.m. – 4:00 p.m.	Do Eco-Communication Strategies Reduce Energy Use and Emissions from Light Duty Vehicles? **Mario Teisl**, **Jonathan Rubin**, and **Caroline L. Noblet**, University of Maine
	4:00 p.m. – 4:30 p.m.	Vehicle Choices, Miles Driven, and Pollution Policies **Don Fullerton**, **Ye Feng**, and **Li Gan**, University of Texas at Austin
	4:30 p.m. – 4:45 p.m.	Discussant: **Ed Coe**, EPA, Office of Transportation and Air Quality
	4:45 p.m. – 5:00 p.m.	Discussant: **Winston Harrington**, Resources for the Future
	5:00 p.m. – 5:30 p.m.	Questions and Discussion
5:30 p.m.	**Adjournment**	

October 18, 2006:

8:45 a.m. – 9:15 a.m.	**Registration**	
9:15 a.m. – 12:20 p.m.	**Session IV: Air Issues** Session Moderator: **Elaine Frey**, EPA, National Center for Environmental Economics	
	9:15 a.m. – 9:45 a.m.	Testing for Dynamic Efficiency of the Sulfur Dioxide Allowance Market **Gloria Helfand**, **Michael Moore**, and **Yimin Liu**, University of Michigan
	9:45 a.m. – 10:05 a.m.	When To Pollute, When To Abate: Evidence on Intertemporal Use of Pollution Permits in the Los Angeles NOx Market **Michael Moore** and **Stephen P. Holland**, University of Michigan

10:05 a.m. – 10:20 a.m.	**Break**	
	10:20 a.m. – 10:50 a.m.	A Spatial Analysis of the Consequences of the SO_2 Trading Program **Ron Shadbegian**, University of Massachusetts at Dartmouth; Wayne Gray, Clark University; and Cynthia Morgan, EPA
	10:50 a.m. – 11:20 a.m.	Emissions Trading, Electricity Industry Restructuring, and Investment in Pollution Abatement **Meredith Fowlie**, University of Michigan
	11:20 a.m. – 11:35 a.m.	Discussant: **Sam Napolitano**, EPA, Clean Air Markets Division
	11:35 a.m. – 11:50 a.m.	Discussant: **Nat Keohane**, Yale University
	11:50 a.m. – 12:20 p.m.	Questions and Discussion
12:20 p.m. – 1:30 p.m.	**Lunch**	
1:30 p.m. – 4:35 p.m.	**Session V: Water Issues** Session Moderator: **Cynthia Morgan**, EPA, National Center for Environmental Economics	
	1:30 p.m. – 2:00 p.m.	An Experimental Exploration of Voluntary Mechanisms to Reduce Non-Point Source Water Pollution With a Background Threat of Regulation Jordan Suter, Cornell University, Kathleen Segerson, University of Connecticut, Christian Vossler, University of Tennessee, and Greg Poe, Cornell University
	2:00 p.m. – 2:30 p.m.	Choice Experiments to Assess Farmers' Willingness to Participate in a Water Quality Trading Market **Jeff Peterson**, Washington State University, and Sean Fox, John Leatherman, and Craig Smith, Kansas State University
2:30 p.m. – 2:45 p.m.	**Break**	
	2:45 p.m. – 3:15 p.m.	Incorporating Wetlands in Water Quality Trading Programs: Economic and Ecological Considerations **Hale Thurston** and Matthew Heberling, EPA, National Risk Management Research Laboratory, Cincinnati, Ohio
	3:15 p.m. – 3:35 p.m.	Designing Incentives for Private Maintenance and Restoration of Coastal Wetlands **Richard Kazmierczak** and **Walter Keithly**, Louisiana State University at Baton Rouge
	3:35 p.m. – 3:50 p.m.	Discussant: **Marc Ribaudo**, USDA, Economic Research Service
	3:50 p.m. – 4:05 p.m.	Discussant: **Jim Shortle**, Pennsylvania State University
	4:05 p.m. – 4:35 p.m.	Questions and Discussions
4:35 p.m. – 4:45 p.m.	**Final Remarks**	

4:45 p.m. **Adjournment**

Testing for Dynamic Efficiency of the Sulfur Dioxide Allowance Market

By

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Abstract: The sulfur dioxide (SO_2) allowance market is the "grand policy experiment" in environmental regulation. Evidence is mixed on the efficiency of the market. We examine the intertemporal allowance market using monthly data on SO_2 spot market prices from late 1994 through 2003. We test whether the price path follows the Hotelling *r*-percent rule for intertemporal arbitrage. This is a direct test of a competitive market equilibrium and, thus, an indirect test of dynamic efficiency. The Hotelling rule is rejected on balance, which provides limited evidence of inefficiency. We also seek to explain the movement of monthly allowance prices. In an environment of uncertainty, structural breaks in allowance price trends and unexpected changes ("shocks") in markets related to the SO_2 market can affect price movements. We include variables for two endogenous breaks and for shocks in five related markets. These variables substantially improve the goodness of fit. These new insights on the SO_2 allowance market are especially relevant as the market is serving as the template for national and international markets in carbon dioxide emissions.

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1 Introduction

The sulfur dioxide (SO_2) allowance market is the "grand policy experiment" in environmental regulation: a large-scale, long-term program to achieve cost-effective regulation of pollution emissions through an economic policy instrument.[1] An "allowance" is a tradable permit under the Clean Air Act's Acid Rain Program. An allowance issued in a particular year authorizes its owner to emit one ton of SO_2 in that year or any subsequent year (under the program's banking provision). Launched in 1994, the market has allocated SO_2 allowances among electricity producers for over a decade. There is widespread agreement about the program's dramatic success (Gayer, Horowitz, and List, 2005), due in part to estimates of compliance cost savings during 1995-99 of $358 million per year (68%) relative to command-and-control regulation (Ellerman et al., 2000). Based on this success, the SO_2 market is serving as the template for CO_2 emission markets under the European Union Emission Trading System (Kruger and Pizer, 2004) and the seven-state Regional Greenhouse Gas Initiative in the northeastern United States (Regional Greenhouse Gas Initiative, 2006).

Yet evidence on the efficiency of the SO_2 market is mixed. Joskow, Schmalensee, and Bailey (1998) conclude that "a relatively efficient private market" had developed by mid-1994 (even prior to the official start of the program), based on evidence that a transparent, single price was clearing the SO_2 market and that intertemporal markets had emerged for allowances of future vintages.[2] Ellerman and Montero (2005) also find evidence of "reasonably efficient banking" of SO_2 allowances over the 1995-2002 period.[3] In contrast, Carlson et al. (2000) find that a large share of potential gains from trade went unrealized in 1995 and 1996, suggesting that a mature market had yet

[1] Stavins (1998) coined the phrase "grand policy experiment." The SO_2 market implements twenty-five years of economic literature that develops the theoretical and policy analysis of tradable permit systems (beginning with Crocker, 1966; Dales, 1968; Montgomery, 1972).
[2] This evidence of market efficiency is also reported in related publications by the same researchers along with additional co-authors; see Ellerman et al. (2000) and Schmalensee et al. (1998).
[3] Ellerman and Montero (2005) develop a theoretical model of efficient banking (storage) of allowances, transform it into a simulation model, and compare simulated banking with actual banking under the program. Their approach to analysis of the intertemporal allowance market differs from ours: they analyze intertemporal quantities, while we analyze intertemporal prices.

to develop. Keohane (2006) finds similar evidence of substantially lower estimated cost savings of $153 million per year during 1995-99, which is only 17% of his estimated abatement costs under command-and-control regulation. What, then, is the status of the market: are profit opportunities being fully exploited in an efficient market? Or, are firms not maximizing profits, perhaps due to institutional barriers? As the grand policy experiment continues, analysis of market efficiency is critical for ongoing evaluation of the SO_2 market and for design of new pollution markets.

Our research addresses two questions about the intertemporal allowance market. First, what explains the intertemporal movement of monthly prices in the SO_2 spot market from August 1994 through December 2003? Initially falling through 1996, spot prices went through several peaks (above $200 per ton) and troughs (below $100 per ton) through 2003.[4] We apply Schennach's (2000) theoretical model of the intertemporal SO_2 allowance market under uncertainty to address this question.[5] Allowances are a storable commodity in the model, so that the risk-free interest rate, an SO_2-specific risk premium, and convenience yield affect SO_2 price movements. As well, according to Schennach, unexpected changes ("shocks") in markets related to the SO_2 market and unanticipated regulations can affect price movements. Our econometric analysis implements this model, including the use of variables for shocks in five markets related to SO_2 (electricity, low-sulfur coal, high-sulfur coal, natural gas, and labor).

Second, is the SO_2 price path consistent with a competitive equilibrium in the intertemporal market? As part of our analysis, we test for equilibrium using the Hotelling rule: whether allowance

[4] Prices climbed dramatically since late 2003 to break the $1,000 per ton barrier in late 2005. This appears to be caused by the Clean Air Interstate Rule, which was proposed in January 2004 and issued formally in March 2005. The rule will significantly reduce the overall cap on SO_2 emissions beginning in 2010. As in Ellerman and Montero (2005) and Liski and Montero (2005), we do not use data after December 2003 because of the structural adjustment in the market due to this rule.
[5] Schennach's (2000) approach follows the literature on nonrenewable resource markets (Hotelling, 1931), storable commodity pricing under uncertainty (e.g., Pindyck, 1993), and bankable pollution permits (e.g., Kling and Rubin, 1997).

prices follow an *r*-percent trajectory over time (Hotelling, 1931).[6] The question of competitive equilibrium relates directly to the issue of SO_2 market efficiency. Under the *first fundamental theorem of welfare economics*, evidence of competitive equilibrium would imply dynamic efficiency (e.g., Mas-Colell, Whinston, and Green, 1995). In this case, dynamic efficiency involves minimizing present-value cost of compliance with the intertemporal SO_2 regulation. While evidence against a competitive equilibrium would not necessarily rule out efficiency, it does appear to create the possibility of arbitrage profits from intertemporal allowance reallocation.

Here we highlight key results on the two questions. First, our ability to explain SO_2 price movements is greatly improved by two sets of variables: variables for structural breaks in the allowance price path and variables for shocks in SO_2-related markets. We employ an econometric method (Lee and Strazicich, 2003) to identify two endogenous structural breaks in the price path in the late 1990's. These breaks correspond to the time period when convenience yields for allowances of future vintages rose dramatically as a percentage of price. Including the structural breaks and the market shocks substantially improves the goodness of fit and, as well, lowers the standard error of the estimated coefficient on the interest rate variable.

Second, the Hotelling rule is rejected regularly in hypothesis tests on the estimated coefficients for the interest rate variable in several model specifications. We conclude that the allowance market was not in a competitive equilibrium during 1994 to 2003. As the first direct econometric evidence on the SO_2 allowance market, this is an important new insight on performance of the market.

The remainder of the paper is organized as follows. Section 2 presents descriptive evidence on the intertemporal allowance market. Section 3 describes the empirical models for the analysis.

[6] Tests of the Hotelling rule for nonrenewable resources are notoriously difficult to implement because marginal extraction cost is rarely observed with accuracy (Berck, 1995). We circumvent this problem with SO_2 allowances because of their costless extraction and storage, which makes the SO_2 case ideal for testing the rule.

Section 4 describes the data, variables, and econometric methods. Section 5 presents the results, and Section 6 provides closing remarks.

2 Evidence from the Intertemporal Allowance Market

Each year, allowances are distributed free of charge to firms that operate coal-fired power plants in the United States.[7] The birth year of an allowance is defined as its *vintage*, for example, an allowance issued this year is vintage 2006. An allowance can either be used to cover a ton of SO_2 emissions in its birth year or be banked (stored) for future use. The fact that a banked allowance is a perfect substitute for an allowance of a future vintage gives rise to the possibility of an intertemporal market.

Two characteristics of the program then created clear incentives for banking and thereby brought the intertemporal market into reality. First, allowance allocations were substantially higher in 1995 and 1996 than originally planned due to a variety of special provisions (Ellerman et al. 2000). Making compliance unexpectedly easier facilitated banking; the highest levels of annual storage occurred in those years (Table 1). Second, allocations to individual electric generating units decreased substantially in Phase II of the program relative to Phase I. (Phase I covered 1995-99, while Phase II covers 2000 and thereafter. Phase I encompassed only the 263 dirtiest large generating units, and Phase II encompasses almost all coal-fired generating units.) Thus, while the aggregate allocation increased in 2000, the per-unit allocations decreased for the dirtiest units. Electricity producers could be expected to ease this transition by banking allowances from Phase I to use in Phase II. This hypothesis is consistent with the pattern of accumulating unused allowances during Phase I, followed by drawdown of the stock during Phase II (Table 1).

A firm with extra allowances in this program has three choices: it can use the allowances itself (by reducing abatement effort or generating more electricity), it can sell the allowances to

[7] Power plants in Hawaii and Alaska are exempt from the regulation.

another source, or it can bank the allowances. In equilibrium under certainty, the present value of an allowance in any of these three uses should be equivalent; otherwise, profits can be made by reallocating allowances to the higher-valued use. The first two choices lead to equating of marginal costs of abatement across sources. The third option leads to the applicability of the Hotelling rule: a firm will be indifferent between current and future use of an allowance if the present value of allowances is the same in the current and future markets, or (put another way) if the undiscounted price path of allowances provides the same return as the best alternative monetary investment. Under uncertainty, though, this simple dynamic price path becomes more complex.

Schennach's (2000) theoretical model of the intertemporal SO_2 allowance market under uncertainty applies the economics of storable commodities (e.g., Pindyck, 1993). It is useful here to fix several ideas from the model about the SO_2 price path. With active banking and a positive balance in storage (i.e., an interior solution), the equilibrium condition for the price path[8] is

$$E_t p_{t+1} = (1 + r_t^f + r_t) p_t - y_t, \quad (1)$$

where E is the expectations operator, subscripts t and $t+1$ denote time, p_t is SO_2 allowance price, r_t^f is the risk-free interest rate, r_t is the SO_2 asset's risk premium (in the spirit of CAPM), and y_t is convenience yield in dollars per ton. Convenience yield is the service flow to holding a stock of a storable commodity in inventory to protect against complete depletion, or a "stockout" (Pindyck, 1993). Uncertainty must be present in the market to create convenience yield.

The remainder of this section presents descriptive evidence on relationships in the intertemporal allowance market, with a focus on convenience yield. We contrast evidence presented by Ellerman et al. (2000, p. 185-190) on the early years of the market (1994-99) with more recent evidence.

[8] Schennach (2000) develops the model in continuous time, yet we present the model in discrete time for empirical purposes.

Ellerman et al. examine the intertemporal allowance market from the perspective of the forward market for future vintages of allowances during 1994-99. They assess two characteristics of these markets: the term structure of the market and the convenience yields on future vintages. The term structure of the forward market is measured by the time horizon of future allowance vintages that were selling in this market. In July 1995, the term structure was relatively short: transactions were occurring on vintages of up to +3 years, i.e., vintages that matured in 1996 through 1998. From January 1996 through January 1999, the term structure lengthened substantially, with transactions of vintages that matured in the range of +6 to +8 years. They conclude that the reasonably long term structure of the forward market in the late 1990's reflected a robust, healthy intertemporal market.

Convenience yields on future vintages reflected even stronger descriptive evidence of a relatively efficient intertemporal market. The relationship among convenience yield and the immediate settlement prices on current and future vintages is a particularly simple way to gauge the workings of an intertemporal market. Ellerman et al. (p. 187) derive this relationship as

$$p_t^{v(y)} = p_t^{v(y+t)} + c_t^{v(y+t)},$$

where $p_t^{v(y)}$ is the price of the current-year vintage at time t, $p_t^{v(y+t)}$ is the price of a $+t$ years vintage at time t, and $c_t^{v(y+t)}$ is the present-value convenience yield on the +t years allowance at time t. With both convenience yield and the CAPM risk premium equal to zero, current and future vintages should have the same price at t. This is a version of the Hotelling rule for the case of costless extraction and storage. With a positive convenience yield, future vintages trade at a discount to the current vintage. Ellerman et al. report relatively small present-value convenience yields for future vintages in the January 1996 through January 1999 period. For example, +3 vintages traded at a one to two percent discount relative to the current vintage, and +7 vintages traded at a three to four percent discount. Based on this evidence, they conclude that a "robust and efficient" intertemporal allowance market had emerged by early 1996.

The intertemporal allowance market appeared to be operating with textbook-quality efficiency in the late 1990's. The convenience yield on the price of a +7 years vintage allowance for July 1998 is the last datum reported by Ellerman et al.; the +7 vintage traded at a discount of about four percent relative to the current vintage at that time. Four percent is similar to the 5.8 percent discount on the +7 vintage from the annual EPA auction in late March of 1998.

Yet the functioning of the intertemporal market changed dramatically during the 1998-99 period. Table 1 presents publicly available data from the annual EPA auction.[9] Through the March 1998 auction,[10] present-value convenience yields on +7 year allowances were consistently small, with +7 allowances trading at a 4.3 to 5.8 percent discount.[11] In the March 1999 auction, however, the +7 year allowance traded at a 16.5 percent discount relative to the current vintage's price, and the absolute value of the convenience yield was also substantially higher than earlier levels, at $33 per ton. The March 2000 auction yielded an even higher convenience yield in percentage terms: a discount of 56.1 percent relative to the current vintage's price. The discounts remained relatively high thereafter and peaked in the 2005 auction at 62.3 percent.

With convenience yield as an indicator, uncertainty in the allowance market appeared to increase substantially prior to the March 1999 EPA auction and even more prior to the March 2000 auction. The first increase in uncertainty corresponds to the period of the first dramatic increase in SO_2 spot market prices, during which price reached a temporary peak at $197 per ton in July 1998 (Figure 1). The second increase in uncertainty – which occurred quite rapidly – corresponds to a

[9] In late March of each year, EPA sells 2.8 percent of the total number of allowances available that year in an auction. The auction data have the advantage of transparency. Actual transaction prices on forward markets are not publicly available from the private brokerage firms. [[Check accuracy of this statement!!]]
[10] Two future vintages, a +6 year vintage and a +7 year vintage, were sold in the 1995-1997 EPA auctions. Beginning 1998 and continuing thereafter, only a +7 year vintage was sold in the auctions.
[11] Early in the program, researchers criticized the discriminatory price mechanismof the EPA auction for resulting in a lower market-clearing price than would occur in a uniform price auction (Cason, 1993, 1995; Cason and Plott, 1996). Ellerman et al. (2000, p. 171) dismiss this criticism, arguing instead that the private allowance market imposes opportunity-cost bounds that effectively transform the auction into a common-value auction.

period of a rapid decrease in spot prices, from over $210 to under $140 per ton. Thus, the structural changes in convenience yield in the forward market are mirrored by volatility in the spot market.

A relevant fact is that these present-value convenience yields grew large at the same time that the stock of stored allowances was peaking (Table 1). For example, information on the end-of-year stock for 1999 would just become public prior to the March 2000 auction. Moreover, the convenience yields remained large in the first several years of Phase II of the program. Information was available during this period that, although the aggregate stock was declining, its rate of decline was much slower than its rate of growth during Phase I. Thus, convenience yields were increasing despite the evidence of substantial potential liquidity in the allowance market. In effect, the market was putting substantial weight on the possibility that the allowance stock could be depleted within the time frame of a +7 vintage allowance.

The descriptive evidence on the intertemporal allowance market has three implications for our analysis. One, the relative stability of the spot and forward markets during 1996-97 has given way to volatility and relatively large convenience yields. A model of the allowance market under uncertainty – not under certainty – thus appears appropriate. Two, the simple story of expected price movements following the Hotelling rule is insufficient in light of the evidence on convenience yields. The approach thus needs to incorporate convenience yield. Three, structural changes may be an important feature of allowance markets, and thus our methodology needs to allow for their occurrence.

3 Empirical Models

Schennach's (2000) theoretical model of the intertemporal SO_2 allowance market guides our empirical approach. The model describes the planner's problem of minimizing discounted SO_2 abatement costs over an infinite time horizon subject to time-dated allowance allocation, use, and storage. The planner's solution is identical to the competitive market equilibrium based on standard decentralization results. With certainty (perfect foresight), the model predicts that the SO_2 price path would increase smoothly at the rate of interest according to the Hotelling rule. The price path, of course, was quite volatile from 1994 through 2003 (Figure 1), so we reject the certainty model in favor of Schennach's model of the market under uncertainty.

With uncertainty, holding an allowance can generate two returns in addition to the interest rate. One is a risk premium (or discount) to holding allowances as an asset in a diversified portfolio of investments. This type of return has been studied extensively using the capital asset pricing model (CAPM).[12] A second return is convenience yield, which was described earlier. The model incorporates these two arguments.

Uncertainty in the SO_2 market may arise due to market, regulatory, or technological uncertainty (Schennach, 2000).[13] The error term in the regression reflects this new information. Yet we also attempt to capture this new information systematically by constructing variables for shocks in markets related to the SO_2 market.

3.1 Base Model

To develop an estimable form of equation (1), we manipulate the algebra and convert r_t to the empirical specification for CAPM to yield an expression for the expectation at *t* of the allowance price at *t+1*:

$$E_t p_{t+1} - p_t = r_t^f p_t + \frac{s_{am}}{s_{mm}}(r_t^m - r_t^f) p_t - y_t, \tag{2}$$

where r_t^m is the rate of return on the market portfolio of risky assets, s_{am} is the covariance between the rate of return on SO_2 allowances and r_t^m, and s_{mm} is the variance of r_t^m. The variable E_t

[12] Gaudet and Khadr (1991) and Slade and Thille (1997) develop models that integrate the Hotelling and CAPM models. The approach used here is consistent with their models. Slade and Thille also apply the model empirically.
[13] Market uncertainty reflects uncertainty in markets related to the SO_2 market, such as the natural gas market. Regulatory uncertainty reflects uncertain future developments in environmental regulation or regulation of electricity markets. Technological uncertainty reflects uncertain future develops in SO_2 abatement technology or "clean coal" technologies.

represents rational expectations conditional on information available at time *t*. The first term on the right-hand side represents the Hotelling rule for cost-minimizing intertemporal arbitrage in the SO_2 market. The second term on the right-hand side is the risk premium for holding SO_2 allowances as part of a diversified portfolio. The expression $(r_t^m - r_t^f)$ is the excess return on the market portfolio at time *t*. The risk premium for holding allowances is positive when σ_{am} is greater than zero, i.e., allowances need to earn a positive premium when the covariance is positive. With risk-averse consumers (investors), an asset return that varies positively with the market portfolio is a liability. The last term on the right-hand side continues as convenience yield.

Because of unexpected shocks to the SO_2 market, the expected value of p_{t+1} is known only with error at time *t*. In other words, the actual price at *t+1* can be written as $p_{t+1} = E_t p_{t+1} + \varepsilon_{t+1}$.[14] The error term ε_{t+1} reflects new information about the SO_2 market that becomes available between *t* and *t+1*. The expected price path is not observable; substituting for $E_t p_{t+1}$ in equation (2) produces an equation with observable arguments:

$$p_{t+1} - p_t = r_t^f p_t + \frac{\sigma_{am}}{\sigma_{mm}}(r_t^m - r_t^f)p_t - \psi_t + \varepsilon_{t+1}. \tag{3}$$

To convert to an econometric model, we assume that convenience yield is constant ($\psi_t = \psi$) and rewrite the equation as

$$p_{t+1} - p_t = \alpha + \beta_1 r_t^f p_t + \beta_2 (r_t^m - r_t^f)p_t + \varepsilon_{t+1}, \tag{4}$$

where $\alpha = -\psi$ and $\beta_2 = \sigma_{am}/\sigma_{mm}$, which is standard practice for CAPM. The restriction $\beta_1 = 1$ tests the Hotelling rule, which is the test for a competitive market equilibrium. The sign and significance of β_2 provides information on the CAPM risk premium for SO_2 allowances. Equation (4) is labeled the *Base Model.*

[14] Mankiw and Summers (1984) state the relation between actual and expected interest rates in this form.

Empirically, the intercept term a represents an average for convenience yield over time. We also incorporate two endogenous structural breaks, the first in February 1998 and the second in September 1999. The breaks are intercept shifters, so that the regression results produce information on averages for convenience yield from three phases of the market: (a) Aug. 1994-Jan. 1998, (b) Feb. 1998-Aug. 1999, and (c) Sept. 1999-Dec. 2003.

3.2 Base Model and Market Shocks

An extension of the *Base Model* puts structure on the new information entering the market between t and $t+1$. Comparison of equations (2) and (4) shows the difference between the *expected* and *actual* SO_2 price paths in an environment of uncertainty. The expected path in (2) evolves according to the equilibrium returns and service flows earned in the market. With traders lacking perfect foresight as in (4), however, the actual price also changes by another term, e_{t+1}, when new information arrives in the market between t and $t+1$. This occurs whenever the resolution of an uncertainty deviates from its expected value.

To capture the role of market uncertainty, we explicitly model new information from unexpected changes in five markets that might affect the SO_2 market.[15] Conceptually, the SO_2 abatement cost function for an electricity producer can be used to identify markets related to the SO_2 market. The arguments of an abatement cost function include: electricity price, low-sulfur coal price, high-sulfur coal price, natural gas price, wage rate, and SO_2 price. The new information from these five markets is derived as forecast errors from time-series models of market prices for low-sulfur coal, high-sulfur coal, natural gas, and labor; and of market quantities for electricity.[16] That is, we forecast monthly prices in each of these markets; compute forecast errors for each market as the

[15] The other general sources of new information – "news" emanating from regulatory and technological uncertainty – are not incorporated into the analysis. As information sources, they are more difficult to model as events that occur at a particular time. Moreover, we conjecture that new information in the five markets incorporates new information from the other sources. For example, new information about a breakthrough in "clean coal" technology should cause an unforeseen change in low- and high-sulfur coal prices.

[16] We use electricity sales instead of prices because prices in electricity markets are still regulated in many places and are not determined only by supply and demand.

difference between actual price and forecast price; and construct five independent variables. The data and time-series models used for this exercise are described further in Section 4.

We develop a second empirical specification using the idea that new information can explain SO_2 price movements. The error term e_{t+1} depends on these five sources of news:

$$e_{t+1} = f(elecusefe_{t+1}, lscprcfe_{t+1}, hscprcfe_{t+1}, ngasprcfe_{t+1}, wagefe_{t+1}) + n_{t+1},$$

where $elecusefe_{t+1}$ is forecast error for electricity sales at $t+1$, $lscprcfe_{t+1}$ is forecast error for low-sulfur coal price at $t+1$, $hscprcfe_{t+1}$ is forecast error for high-sulfur coal price at $t+1$, $ngasprcfe_{t+1}$ is forecast error for natural gas price at $t+1$, $wagefe_{t+1}$ is forecast error for wage rates at $t+1$, and n_{t+1} is the unexplained error term at $t+1$. Substituting this expression for e_{t+1} into equation (4) and converting to an estimable form yields

$$\begin{aligned} p_{t+1} - p_t = a + b_1 r_t^f p_t + b_2 (r_t^m - r_t^f) p_t + b_3 elecusefe_{t+1} + b_4 lscprcfe_{t+1} + \\ b_5 hscprcfe_{t+1} + b_6 ngasprcfe_{t+1} + b_7 wagefe_{t+1} + n_{t+1}. \end{aligned} \tag{5}$$

Equation (5) is labeled the *Base Model and Market Shocks*. Its goal is to explain the volatile nature of SO_2 spot market prices.

4 Data, Variables, and Econometric Methods

In preparation for estimation of equations (4) and (5), variables are constructed using monthly data from August 1994 through December 2003, which totals to 113 observations. The SO_2 spot price (p_t, in dollars per ton) is the monthly Market Price Index from Cantor Environmental Brokerage. Cantor's index series is the most widely cited source of data on SO_2 prices. The U.S. Environmental Protection Agency reports this series in official publications, and it has been used in earlier research (e.g., Joskow, Schmalensee, and Bailey, 1998). The risk-free rate of return (r_t^f, in percentage points at monthly rates) is the 3-month Treasury bill.[17] The rate of return on the market portfolio of risky assets (r_t^m, in percentage points at monthly rates) is the daily average S&P 500 Price Index for a given month. The appendix describes the sources of these data. Table 2 reports summary statistics for the variables used to estimate equation (4), $p_{t+1} - p_t$, $r_t^f p_t$, and $(r_t^m - r_t^f)p_t$.

Equation (5) incorporates the variables for new information on prices in five markets that are related to the SO_2 market through the SO_2 abatement cost function. The markets related to the SO_2 market are: electricity sales, low-sulfur coal price, high-sulfur coal price, natural gas price, and wage in the public utilities and transportation sector. The five variables are forecast errors from monthly predictions of each series. Three steps are followed to produce these variables. First, we estimate a model (an ordinary least squares regression including a time trend and monthly dummies) to forecast each series using monthly data that begins in January 1988 (or January 1990 for electricity sales). These data pre-date the formation of the SO_2 market. Second, we apply the model to forecast the series for every month in our study period (August 1994 through December 2003). The forecasts use data from all months prior to the month at hand to produce the forecast for that month. Thus, for each data series, we generate 112 regressions and 112 predictions spanning September 1994 to December 2003. (We term this procedure the "one-step-ahead" forecast.) Third, we compute the forecast error as the difference between actual value and forecast value for every month of the study. This creates a measure of new information, or a shock, emanating from each of the five markets. Further detail on this method is in the Appendix.

The forecast models are estimated with monthly data. Electricity sales data are from the Energy Information Administration and are measured in megawatt-hours. Low-sulfur coal, high-sulfur coal, and natural gas prices are from the Federal Energy Regulatory Commission and are in cents per million BTUs. Wage rates for public utility and transportation labor are from the Bureau of

[17] The 3-month Treasury bill is the instrument of shortest duration for which monthly data exist for the study period. Monthly data for the 1-month Treasury bill are not available for this period.

Labor Statistics in dollars per hour. The interest rate data, from the Federal Reserve, are monthly data expressed as annual percentages; we convert those annual percentages to monthly percentages. The appendix also describes these data in more detail.

Using these data and the forecast models, five variables are constructed for use in estimating equation (5): $elecusefe_{t+1}$, $lscprcfe_{t+1}$, $hscprcfe_{t+1}$, $ngasprcfe_{t+1}$, and $wagefe_{t+1}$. As a robustness check, we also consider the possibility that new information might not be dispersed immediately and, instead, it affects the SO_2 market with a time lag. This is an empirical conjecture without a formal basis in theory. The implication is that the shock variables at time t ($elecusefe_t$, $lscprcfe_t$, $hscprcfe_t$, $ngasprcfe_t$, and $wagefe_t$) affect the SO_2 price change at time $t+1$ ($p_{t+1} - p_t$). Table 2 reports the summary statistics for the time $t+1$ version of these variables; the statistics for the time t version are very similar.

A second robustness check incorporates variables from the Arbitrage Pricing Theory (APT) as potential influences on SO_2 allowance price movements. The APT, as derived in the finance literature, incorporates macroeconomic factors as potential influences on asset price (Cambell, Lo, and MacKinlay, 1997). Following Slade and Thille (1997),[18] variables are developed for the forecast errors of three macroeconomic factors: the Consumer Price Index ($CPIfe_{t+1}$), the interest rate on the 10-year Treasury bond ($10yrbondfe_{t+1}$), and the Industrial Production Index ($IPIfe_{t+1}$). To compute forecast errors, we use the same methods as described above for the market shock variables. The appendix describes the data for the macroeconomic factors, and Table 2 reports the summary statistics for their forecast errors.

Because econometric results may be unreliable if the dependent variable is nonstationary, we first need to test the stationarity of allowance prices and their first difference. One of the possible

[18] Slade and Thille (1997) integrate the Hotelling model of nonrenewable resource markets with the CAPM and APT models from the finance literature. They study shadow price movements in Canadian copper mines.

complications of unit root tests for stationarity is that the presence of structural changes during the time series may make rejection of a unit root more difficult (Perron, 1989). In the time period under study here (August 1994 – December 2003), a number of events occurred that may have created structural changes.[19] As a result, we use a method developed by Lee and Strazicich (2003) that endogenously looks for structural breaks while testing for the existence of a unit root. This method is preferable to including all possible structural shifts in our model, since the latter would require significant assumptions about when the possible shifts first affected the market and would lead to many fewer degrees of freedom. Using this method, we are not able to reject the presence of a unit root for allowance prices, but we are able to reject the presence of a unit root for the first difference of allowance prices.[20] We use the latter as the dependent variable in the regression models.

An advantage of this method, as noted, is that the data themselves suggest the possible timing of structural breaks. Lee and Strazicich include two methods for the test (one with up to two shifts in level, one with up to two shifts in both level and trend), and we conduct the test both for data through 2003 and for data through 2004 (to check for shifts late in the dataset). Based on these results, we develop a candidate list of dates for structural breaks in the model: March 1997, February 1998, September 1999, October 2000, and April 2003. We include these as dummy variables in our estimation of equations (4) and (5). Only the breaks in February 1998 and September 1999 are statistically significant. We drop the others from the model.

One concern with the shock variables is a potential endogeneity problem with the price shocks for low- and high-sulfur coal. This is addressed with a Hausman test for endogeneity bias. We developed several instruments for the two coal price shocks; these include Btu content of low-

[19] These include a change in the president, proposed and actual regulatory changes (e.g., proposed revisions to New Source Review and changes in regulation of particulate matter), legal decisions (including rulings on national ambient air quality standards for ozone), negotiations over international greenhouse gas controls, and disruptions in the California energy market. Indeed, we stop our series at December 2003 because the Clean Air Interstate Rule, proposed in January 2004, may have contributed to sudden major movements in the allowance market.
[20] These results were consistent with the results of an augmented Dickey Fuller test on the two series.

sulfur coal, Btu content of high-sulfur coal, ash content of low-sulfur coal, ash content of high-sulfur coal, a rail cost adjustment factor (RCAF), RCAF squared, and total coal consumption in industry.[21] The appendix describes the data for these variables. We execute the Hausman test following procedures defined in Wooldridge (2002), which allows for generation of Newey-West standard errors. We could not reject the null hypothesis of exogeneity.

We estimate equations (4) and (5) using OLS. To account for the possibility that the error terms (e_{t+1} or n_{t+1}) may be serially correlated and heteroskedastic, we apply the Newey-West procedure to generate robust standard errors.[22]

5 Results

We estimate the *Base Model* of equation (4) with two variations: with and without the endogenous structural breaks. The first break, *break1*, is a dummy variable equal to 1 in February 1998 and 0 thereafter. The second break, *break2*, is a dummy variable equal to 1 in September 1999 and 0 thereafter. Similarly, we estimate the *Base Model and Market Shocks* of equation (5) with and without the structural breaks. The results are reported in Table 3. Section 5.2 reports robustness checks to several additional specifications of the model.

5.1 General Results

One question is: Do allowance prices follow an r-percent trajectory over time (the Hotelling rule)? We address this first since the answer is relatively compact. For the Hotelling hypothesis to be maintained, the null hypothesis is that $b_1 = 1$. The estimated coefficients (b_1) for the interest rate variable ($r_t^f p_t$) are negative and of similar magnitude across the four specifications. In the most parsimonious specification (*Base Model* without structural breaks), the null hypothesis cannot be

[21] We thank Nat Keohane for insight into the coal market.
[22] We specify twelve lags in the procedure due to the use of monthly data (Wooldridge, 2003).

rejected (p-value = 0.239 in an F test).[23] However, the estimates of the coefficient become more efficient as more control variables are added to the specification. In the *Base Model* with breaks, the null hypothesis also cannot be rejected (p-value = 0.113 in F test), although this result provides very little evidence in favor of the null hypothesis given the p-value. In contrast, the null hypothesis is rejected in the two specifications of *Base Model and Market Shocks*. Without breaks, the null hypothesis is rejected at the 5% level (p-value = 0.039 in F test). With structural breaks, the null hypothesis is rejected at the 1% level (p-value = 0.000 in F test).

On balance, the statistical evidence rejects the Hotelling rule. The *Base Model* without structural breaks does not reject the Hotelling rule, but neither does it provide much confidence that price is rising with the interest rate. With added controls, the estimated coefficient for $r_t^f p_t$ is significantly different from one and the conclusion becomes clear. By rejecting the Hotelling rule, the SO_2 price path is not consistent with a competitive equilibrium in the intertemporal market.

The second general question is: How do the alternative specifications and the variables perform in explaining allowance price movements? The most parsimonious specification (*Base Model* without breaks) represents the essential theory of a storable commodity under uncertainty.[24] It explains only two percent of the variation in allowance price changes (R^2 = 0.02). After including the structural breaks to account for convenience yield, the regression explains nine percent of the variation. The R^2 increases to 25 percent after incorporating both breaks and price-shock variables in the *Base Model and Market Shocks*. Thus, augmenting the theory-derived variables with empirically motivated variables was a useful effort.

At the same time, substantial variation in allowance price movements remains unexplained. Traders apparently were making decisions with information beyond that captured in our analysis. This reflects the complexity of markets in the real world.

[23] The results reported in this paragraph are computed using Newey-West standard errors.
[24] This reflects the model of Gaudet and Khadr (1991).

Among individual variables, the estimated coefficient for the interest rate variable ($r_t^f p_t$) is significantly different from zero at the 1% level in the *Base Model and Market Shocks* with the structural breaks. The estimate, -13.20, suggests that a one unit increase in $r_t^f p_t$ results in a $13.20 decrease in the SO_2 allowance price movement $p_{t+1} - p_t$. As noted above, this price decrease violates the Hotelling rule.

As an asset, SO_2 allowances appear not to be earning a risk premium: the coefficient on the CAPM variable ($(r_t^m - r_t^f) p_t$) is insignificant. This is not surprising; as a relatively new market, there is little experience in understanding its relationship to other investment markets, so investors are unlikely to be holding allowances on a widespread basis.

Based on the theoretical model, we interpret the estimated intercept as average convenience yield during August 1994 through January 1998. The intercept is never statistically significant in these regressions, which suggests that convenience yield was zero for the first several years of the market. In the EPA auction results (Table 1), discounted convenience yield on the +7 year vintages ranged between $3 and $7 per ton in these same years. These numbers suggest quite small convenience yields on the current year vintages. Thus, the regression estimates and auction results are generally consistent.

The estimated coefficients on *break1* estimate the change in average convenience yield that occurred in the second phase, February 1998 through August 1999. The coefficients are slightly over 8 and significant. These imply a decrease in average convenience yield during the second phase relative to the first phase (recall that $a = -y$). The regression estimates from the spot market are inconsistent with the auction results, as convenience yield on the +7 year vintage increased markedly between 1998 and 1999. This is a short phase of 19 months, however, so data points from two auctions might not represent an underlying monthly trend.

Finally, the estimated coefficients on *break2* estimate the change in average convenience yield in the third phase, September 1999 to December 2003. In the *Base Model and Market Shocks* with the structural breaks, the estimated coefficient is about -12 and is significantly different from zero at the 1% level. This implies an increase in average convenience yield of $12 per ton during the third phase relative to the second phase. In comparison, convenience yields on the +7 year vintage allowances increased dramatically in the 2000-03 EPA auctions. In qualitative terms, then, the regression and auction results are consistent during the third phase.

Two of the five variables for market shocks are significant—natural gas price and wage. Their signs suggest that these shocks have a positive effect on the magnitude of SO_2 price movements. The positive influence of the natural gas shock makes sense given that natural gas and SO_2 emissions are substitutes: unexpected increases in natural gas prices, for example, would increase demand for allowances and thus cause an increase in allowance price. We did not have strong priors on the variable for wage shocks. Shock variables for low-sulfur coal price, high-sulfur coal price, and electricity use do not individually affect allowance price movements. Tests of the joint hypothesis that the three variables, together, are significant could not reject the null; they also do not exert a collective influence on price movements. More research is required to understand how new information from electricity and coal markets influences the SO_2 allowance market.

5.2 Robustness Checks

We investigate the robustness of the results to a variety of alternative specifications. One question is whether the new information embodied in the shock variables affects the allowance market with a lag. The five variables for forecast error are similar in magnitude and significance in the new specification—at time *t*—as those for forecast error at time *t+1* (Table 4). Information thus is entering the market both with a lag and concurrently, and the same two related markets (natural gas and labor) are affecting the allowance market. [Here, we need to compute the simple correlation between t and $t+1$ forecast errors to assess whether intertemporal correlation in forecast errors is driving this result.] The estimated coefficients on the interest rate variable ($r_t^f\, p_t$) and the structural breaks (*break1*, *break2*) are also similar in magnitude and significance between the two specifications of forecast-error variables. In particular, the estimated coefficients on the interest rate variable continue to be negative and significantly different from zero. They also are significantly different from one in the test of the Hotelling rule.

A second robustness check comes through inclusion of three variables for macroeconomic shocks, in accordance with the Arbitrage Pricing Theory and prior research on a nonrenewable resource market (Slade and Thille, 1997). These variables—$CPIfe_{t+1}$, $10yrbondfe_{t+1}$, and $IPIfe_{t+1}$—are included in specifications with the base model and market shocks (Table 5). The macroeconomic-shock variables tend not to influence SO_2 allowance price movements. Two exceptions occur: the estimated coefficients on $CPIfe_{t+1}$ and $IPIfe_{t+1}$ are significantly different from zero ($p<0.10$), each in one specification. [[Note: need a joint test of significance of the macroeconomic variables.]] The estimated coefficients on the remaining variables continue their consistent pattern of sign and significance. For example, the coefficients on the interest rate variable are similar in magnitude to earlier specifications, and they are significantly different from both zero and one ($p<0.01$). The forecast-error variables for natural gas prices and wage rates continue to influence allowance price movements.

[[*Note to Discussant: These are the main robustness checks. We still need to report a few other (minor) checks, but won't get to them in the paper and likely won't report them at the conference.*]]

6 Conclusion

The SO_2 allowance market provides a straightforward test of the Hotelling prediction that, with costless extraction, price of a nonrenewable resource increases at the rate of interest over time. Instead, spot market prices were quite volatile—fluctuating in a band roughly between \$100 and \$200 per ton—through 2003. Experts argue that spot market prices were influenced by a combination of regulatory rulings on air pollution emissions and adjustments in related markets (e.g., Burtraw et al., 2005). Schennach (2000) provides a theoretical examination of the SO_2 allowance market under uncertainty and argues for the Hotelling price path after controlling for these shocks. This paper has implemented Schennach's theoretical model in an empirical analysis of the SO_2 allowance price path.

The major finding relates to a competitive equilibrium in the market. We test for the Hotelling rule as the key element of a competitive equilibrium and find evidence, on balance, against the rule. Instead of prices increasing over time, the preponderance of the evidence suggests a downward trend, after controlling for structural changes and market shocks. This evidence suggests that the market is inefficient, with arbitrage profits remaining to be earned. The finding also could lead to an investigation of market power as a source of imperfect competition in the market. On this topic, however, Liski and Montero (2005) find that the behavior of the four largest firms in the market was consistent with perfect competition during 1995 to 2003.[25] Other possible explanations for this inefficiency include: lack of experience in this market; a strong desire to hold allowances to avoid possibilities of future stock-outs; or the (presumably small) opportunities for profits might be less than the costs of finding those profit opportunities.

[25] Liski and Montero (2005) measure firm size according to allowance allocations. In reaching the conclusion of perfectly competitive behavior, they evaluate the pattern of allowance allocations and SO_2 emissions of the four largest firms from 1995-2003 against predictions of their theoretical model of market power in a storable commodity market.

The main empirical innovation of the research is the use of two statistical methods to construct variables to better explain SO_2 allowance price movements. Using time series models, we developed variables for unexpected shocks in markets related to the allowance market. Based on a method for improving unit root tests, we also incorporated variables for two endogenous structural breaks in allowance price movements. These variables substantially improved goodness of fit for the regression equation.[26] At the same time, substantial variation in allowance price movements remains unexplained. As a market created by a government regulation, regulatory uncertainty may influence the market inordinately. Additional research is needed to further explore the influence of regulatory uncertainty on this market.

The SO_2 cap-and-trade program defines a new paradigm for environmental regulation. Its key features are being replicated by several important programs and proposals in the domain of climate policy and air pollution policy. Our research shows that—despite its obvious successes—important questions remain on the performance of the SO_2 allowance market.

[26] Our finding that the endogenous structural breaks improve goodness of fit is similar to the finding by Lee, List, and Strazicich (2006) that inclusion of such breaks improves forecast accuracy of time trends in nonrenewable resource prices.

References

Berck, Peter. "Empirical Consequences of the Hotelling Principle." In Daniel W. Bromley, ed., *The Handbook of Environmental Economics*. Oxford, UK: Blackwell, 1995, 202-221.

Burtraw, Dallas, David A. Evans, Alan Krupnick, Karen Palmer, and Russell Toth. "Economics of Pollution Trading for SO_2 and NO_x." Discussion Paper 05-05, Resources for the Future, Washington, D.C., March 2005.

Busse, Meghan, and Nathaniel O. Keohane. "Market Effects of Railroad Regulation: Coal, Railroads, and the 1990 Clean Air Act." Working paper, December 7, 2005.

Campbell, John Y., Andrew W. Lo, and A. Craig MacKinlay. *The Econometrics of Financial Markets*. Princeton, NJ: Princeton University Press, 1997.

Carlson, Curtis, Dallas Burtraw, Maureen Cropper, and Karen L. Palmer. "Sulfur Dioxide Control by Electric Utilities: What are the Gains from Trade?" *Journal of Political Economy* 108(6) (2000): 1292-1326.

Crocker, Thomas D. "The Structuring of Atmospheric Pollution Control Systems." In Harold Wolozin, ed. *The Economics of Air Pollution*. New York: Norton, 1966.

Dales, J.H. *Pollution, Property and Prices: An Essay in Policy-Making and Economics*. Toronto: University of Toronto Press, 1968.

Ellerman, A. Denny, Paul L. Joskow, Richard Schmalensee, Juan-Pablo Montero, and Elizabeth M. Bailey. *Markets for Clean Air: The U.S. Acid Rain Program*. Cambridge: Cambridge University Press, 2000.

Ellerman, A. Denny, and Juan-Pablo Montero. "The Efficiency and Robustness of Allowance Banking in the U.S. Acid Rain Program." Working Paper 2005-05, Center for Energy and Environmental Policy Research, Massachusetts Institute of Technology, March 2005.

Gaudet, Gérard, and Ali M. Khadr. "The Evolution of Natural-Resource Prices under Stochastic Investment Opportunities: An Intertemporal Asset-Pricing Approach." *International Economic Review* 32 (1991): 441-55.

Gayer, Ted, John Horowitz, and John A. List. "When Economists Dream, They Dream of Clear Skies." *The Economists' Voice* 2 (2005): Article 7.

Hotelling, Harold. "The Economics of Exhaustible Resources." *Journal of Political Economy* 39(2) (April 1931): 137-175.

Joskow, Paul L., Richard Schmalensee, and Elizabeth M. Bailey. "The Market for Sulfur Dioxide Emissions." *American Economic Review* 88(4) (September 1998): 669-685.

Keohane, Nathaniel O. "Cost Savings from Allowance Trading in the 1990 Clean Air Act: Estimates from a Choice-Based Model." In Jody Freeman and Charles D. Kolstad, eds., *Moving to*

Markets in Environmental Regulations: Lessons from Thirty Years of Experience. New York: Oxford University Press, 2006.

Kling, Catherine, and Jonathan Rubin. "Bankable Permits for the Control of Environmental Pollution." *Journal of Public Economics* 64 (1997): 101-115.

Krautkraemer, Jeffrey A. "Nonrenewable Resource Scarcity." *Journal of Economic Literature* 36(4) (December 1998): 2065-2107.

Kruger, Joseph A., and William A. Pizer. "Greenhouse Gas Trading in Europe: The New Grand Policy Experiment." *Environment* 46(8) (2004): 8-23.

Lee, Junsoo, and Mark C. Strazicich. "Minimum LM Unit Root Test with Two Structural Breaks." *Review of Economics and Statistics* 85(4) (2003): 1082-1089.

Lee, Junsoo, John A. List, and Mark C. Strazicich. "Non-renewable Resource Prices: Deterministic or Stochastic Trends?" *Journal of Environmental Economics and Management* 51(3) (2006): 354-370.

Liski, Matti, and Juan-Pablo Montero. "Market Power in a Storable-Good Market: Theory and Application to Carbon and Sulfur Trading." Working Paper 05-016, Center for Energy and Environmental Policy Research, Massachusetts Institute of Technology, November 2005.

Mankiw, N. Gregory, and Jeffrey A. Miron. "The Changing Behavior of the Term Structure of Interest Rates." *Quarterly Journal of Economics* 101(2) (May, 1986): 211-228.

Mankiw, N. Gregory, and Lawrence H. Summers. "Do Long-Term Interest Rates Overreact to Short-Term Interest Rates?" *Brookings Papers on Economic Activity* 1984(1) (1984): 223-247.

Mas-Colell, Andreu, Michael D. Whinston, and Jerry R. Green. *Microeconomic Theory*. Oxford: Oxford University Press, 1995.

Montgomery, W. David. "Markets in Licenses and Efficient Pollution Control Programs." *Journal of Economic Theory* 5(3) (1972): 395-418.

Perron, Pierre. "The Great Crash, the Oil Price Shock, and the Unit Root Hypothesis." *Econometrica* 57 (1989): 1361-1401.

Pindyck, Robert S. "The Present Value Model of Rational Commodity Pricing." *The Economic Journal* 103(418) (1993): 511-530.

Regional Greenhouse Gas Initiative. <www.rggi.org>, accessed August 21, 2006.

Schennach, Susanne M. "The Economics of Pollution Permit Banking in the Context of Title IV of the 1990 Clean Air Act Amendments." *Journal of Environmental Economics and Management* 40(3) (November 2000): 189-210.

Schmalensee, Richard, Paul L. Joskow, A. Denny Ellerman, Juan Pablo Montero, and Elizabeth M.

Bailey. “An Interim Evaluation of Sulfur Dioxide Emissions Trading.” *Journal of Economic Perspectives* 12(3) (Summer 1998): 53-68.

Slade, Margaret E., and Henry Thille. “Hotelling Confronts CAPM: A Test of the Theory of Exhaustible Resources.” *Canadian Journal of Economics* 30(3) (August 1997): 685-708.

Stavins, Robert N. “What Can We Learn from the Grand Policy Experiment? Lessons from SO_2 Allowance Trading.” *Journal of Economic Perspectives* 12(3) (Summer 1998): 69-88.

U.S. Environmental Protection Agency. “Acid Rain Program Annual Report,” 1995-2005 issues, Clean Air Markets Program, http://www.epa.gov/airmarkets/cmprpt/index.html, 2006a. Accessed October 20, 2006.

U.S. Environmental Protection Agency. “Acid Rain Program Allowance Auctions,” 1995-2005 results, Clean Air Markets Program, http://www.epa.gov/airmarkets/auctions/index.html, 2006b. Accessed October 20, 2006.

Williams, Jeffrey C., and Brian D. Wright. *Storage and Commodity Markets*. Cambridge: Cambridge University Press, 1991.

Wooldridge, Jeffrey M. *Econometric Analysis of Cross Section and Panel Data*. Cambridge, MA: The MIT Press, 2002.

Wooldridge, Jeffrey M. *Introductory Econometrics: A Modern Approach*. Mason, OH: Thomson South-Western, 2003.

Appendix: Data, Market Shocks, and Endogenous Structural Breaks

This appendix contains more detailed information on (1) the data used here, (2) the method used to develop the market shock variables, and (3) the method for endogenous determination of structural breaks.

The Data

The data collected and used in this analysis include: prices of SO_2 allowances; data used to develop shocks; data to develop the CAPM and APT variables; and instruments for high- and low-sulfur coal prices for two-stage least squares regression. All the data series run through December 2003.

For SO_2 allowances, the data run from the start of the market (August 1994). For variables that we used to develop shocks (electricity sales; prices of high- and low-sulfur coal; prices of natural gas; hourly wages in the utility sector; interest rates for the 3-month Treasury bill, the prime rate and the 10-year Treasury Constant Maturity Rate; the Standard and Poor's (S&P) 500 stock index; Industrial Production Index; and Consumer Price Index), we collected observations starting from January, 1988 (except for electricity sales, where the data prior to January, 1990, were not available). We started the data series for these variables at this date so that we could estimate the time series models (see below) with data prior to the start of the program. There is a balance, in choosing the length of the data series, between having more data and facing an increased likelihood of structural changes in the series. The choice of 1988 as the initial year seemed to fit that balance. Since Alan Greenspan was appointed to be chair of the Federal Reserve System in 1987, this period can be considered to have a relatively stable monetary regime.

Data used in the two-stage least square analysis (the ash content and Btu content in high- and low-sulfur coal; the rail cost adjustment factor; and total coal consumption by industrial sector) run from August 1994 to December 2003.

Prices of SO2 allowances
Prices of the SO_2 allowances come from the Cantor-Fitzgerald Environmental Brokerage, http://www.emissionstrading.com/. They are measured in dollars per allowance, where an allowance is for one ton of SO_2.

Data used to develop shock variables
The profit function for a firm shifts in response to changes in input prices. To model those changes in input prices, we develop estimates of the forecast error between expected and actual input prices for low-sulfur and high sulfur coal, natural gas, and wages. For the electricity market, we use electricity sales instead of electricity prices as the basis for the shock. Many electricity prices are set in regulated markets and, thus, do not reflect underlying demand and supply fundamentals.

Electricity sales--Electricity sales come from the Energy Information Administration of the U.S. Department of Energy, Form EIA-826 Monthly Electric Utility Sales and Revenue Data, which is found at http://www.eia.doe.gov/cneaf/electricity/page/eia826.html. The value is monthly total electric utility sales measured in megawatt-hours.

Wage data--The wage data are average hourly earnings of production workers for the transportation and public utilities sector from the Bureau of Labor Statistics. They are found at http://data.bls.gov/cgi-bin/srgate, using series ID number CEU4422000006 for the Utilities sector.

Interest rate data--The interest rates for 3-month constant maturities Treasury bonds are found at: http://www.federalreserve.gov/releases/h15/ and http://www.federalreserve.gov/releases/h15/data/Monthly/H15_TCMNOM_M3.txt

The prime rate data come from the website of the St. Louis Federal Reserve Bank, at http://research.stlouisfed.org/fred2/series/MPRIME/117 .

The data for the 10-year constant maturities Treasury bonds are found at: http://research.stlouisfed.org/fred2/data/GS10.txt

All three interest rates are presented as annual percentages, with monthly data frequency. To convert each monthly observation to a monthly interest rate, we used the following formula: if r is the monthly interest rate, and i is the annual interest rate, then $i = (1 + r)^{12} - 1$. Rearranging this formula yields

$$r = \exp^{\left(\frac{\ln(1+\frac{i}{100})}{12}\right)} - 1.$$

S&P 500 Index values--Monthly S&P 500 values are from the daily values at this website: http://www2.standardandpoors.com/spf/xls/index/500_20051224_GALLTOT.xls The daily values are averaged for each month.

Industrial Production Index--The data can be found at http://alfred.stlouisfed.org/series/downloaddata?seid=INDPRO&rid=13 . The chosen vintage date was October 1, 2005 (2005-10-01). It provides the index with year 1997 = 100.

Consumer Price Index--The Consumer Price Index for All Urban Consumers is found at: http://data.bls.gov/cgi-bin/surveymost?cu , choosing "U.S. All items, 1982-84=100 - CUUR0000SA0" in the list.

Prices for High- and Low-Sulfur Coal and Natural Gas--Prices of coal and natural gas are from Form 423 Annual Data, issued by the Federal Energy Regulatory Commission (FERC) (http://www.ferc.gov/docs-filing/eforms/form-423/data-annual.asp#skipnavsub), which gives the cost of coal and gas delivered to electric utilities. The price of each kind of coal as well as natural gas is a quantity-weighed average cost measured as cents per million BTU. Data starting in January 2003 were provided directly by Stephen Scott of the Energy Information Administration from the "FERC-423/EIA-423 Survey Information," rather than from EIA website (where they were not yet available). We use spot market prices, not contract prices. For data prior to August, 1994, we use all plants, since price expectations could be expected to arise from all plants. Between August, 1994, and 2000, we use data only from Table A plants (that is, those plants participating in the SO_2 allowance market); after 2000, we use all plants. The list of Table A plants is from http://www.epa.gov/air/caa/caa404.txt.

Low- and high-sulfur coal must be distinguished when constructing price variables. Carlson, et al. (p. 1321) distinguish low- and high-sulfur coal by whether the coal has sulfur content that would produce more or less than 1.2 pounds of SO_2 per million BTU of heat input; we use the same dividing line. Because the FERC form provides sulfur content, not SO_2 content, we convert sulfur to SO_2 content using the following procedure.

Two conversion factors (1.91 for bituminous coal, and 1.76 for sub-bituminous coal) are used here to convert from sulfur (S) to sulfur dioxide (SO_2) (Nathaniel Keohane, personal communication). We use three types of coal (bitumen, bituminous, and sub-bituminous), but exclude anthracite and lignite when separating coal into the low-sulfur and high-sulfur categories since we lack conversion factors for those types. Anthracite is 0.104% of total tons of coal, while lignite is 9.35%.

The following formula computes sulfur dioxide content using delivery-specific, plant-level data from FERC Form 423:

$$\frac{\text{Sulfur content}}{\text{Heat content}} * 10{,}000 * \text{conversion factor appropriate for the coal} = \text{pounds } SO_2/\text{mmBtu}.$$

Coal with over 1.2 pounds SO_2/mmBtu was considered high-sulfur coal, with the rest low-sulfur coal.

Instruments for the prices of high- and low- sulfur coal
Because of concerns about possible endogeneity of high- and low-sulfur coal prices, we sought variables that would contribute to explanation of coal prices but that are unrelated to the other variables in our regressions. We chose seven variables as instruments: ash content and Btu content from both high-sulfur coal and low-sulfur coal; the rail cost adjustment factor (RCAF), which is an index of railroad costs; RCAF squared; and total coal consumption in industry (excluding commercial, transportation, and energy sector consumption).

Btu content and ash content--Information on ash content and Btu content came from the same Form 423 Annual Data used for high- and low-sulfur coal prices. For both Btu content and ash content, these are quantity-weighted averages measured as Btu per pound and percent by weight, respectively.

Rail cost adjustment factor--The rail cost adjustment factor (RCAF) is an index of the costs of rail shipping. RCAF data were provided by the Association of American Railroads (A. Clyde Crimmel, Jr., personal communication). The RCAF data are restated to a 2002 Q4 =100 base.

Total coal consumption in industry--The coal consumption data are from Table 6.2 of the Monthly Energy Review at Energy Information Administration, found at http://tonto.eia.doe.gov/FTPROOT/monthlyhistory.htm . The data are thousands of short tons of coals consumed by the industrial sector. The data were input manually from the “industrial total” column.

<u>Variables for Shocks</u>

We develop shocks for all the prices expected to influence the price of SO_2 allowances through the cost function for abatement: sales of electricity, high- and low-sulfur coal prices, natural gas price, and wages. In addition, we estimate shocks for the variables related to the Arbitrage Pricing Theory: the interest rate on the 10-year constant maturity Treasury bond, the S&P 500 Index, and the Industrial Production Index. The shocks used in the regressions are the differences between true values and predicted values (true values - predicted values). To calculate the shocks, we need predicted values starting from August, 1994, for the relevant data.

Initially, we developed ARIMA models for each variable. The best-fit models tended to be complex, and they produced variables that performed poorly in explaining SO_2 price movements. Since we are trying to estimate how people in the markets would predict price trends, complex formulations seem unrealistic. We instead use, for each variable, a simple linear model of a time trend and monthly dummies. These models produce variables that perform much better in explaining SO_2 price movements.

We use a method (termed "one-step ahead") of using the data to estimate the predicted values. The coefficients for the model were re-estimated every month and used to provide the prediction for the next month. This model reflects an environment with full information. For most of the coefficients for most of the variables, the coefficients of variation for the coefficients were less than one, suggesting that the time series models were indeed fairly stable.

We also experimented with a second and third method of computing predicted values. The one-step-ahead method performed best, yet the other results are reported as robustness checks. In the second method, using what we term the "short" dataset, we used data only from before the beginning of the SO_2 allowance program – from January 1988 (January 1990 for electricity data) to July 1994 – to estimate the model. This method assumes a very naive form of expectations: the model would not be updated at all.

The third method used the "long" dataset – that is, the data from January 1988 (January 1990 for electricity data) through September 2004 to estimate the model. (We collected data through September 2004 for all variables; only after examining the econometric results did we reconsider use of data for 2004. We did not re-calculate the shocks at that point.) The assumption underlying this model is that the time series model is stable for the whole time period. It has the characteristic of using data from after almost all the predictions for those predictions; this could be considered a disadvantage.

Endogenous Determination of Structural Breaks

Lee and Strazicich (2003) describe a method to determine structural breaks endogenously from time series data. We use their GAUSS computer code to conduct a unit root test and to find structural breaks, both for SO_2 allowance prices and for the first difference of SO_2 allowance prices. Their GAUSS codes can be found at http://www.cba.ua.edu/~jlee/gauss/LStwo.txt . Their Model A includes two changes in intercept for the time series; their Model C includes two changes in intercept and two changes in slopes.

We cannot reject the presence of a unit root for SO_2 allowance prices, though we can reject a unit root for the difference of SO_2 allowance prices.

Following Lee and Strazicich's methods, for SO_2 allowance prices, we identify possible breaks in February 1998 and September 1999 from model C; and breaks in March 1998 and October 2000 from model A. For the difference of SO_2 allowance prices, we identify breaks in March 1997 and June 1998 from model C; and February 1998 and August 1998 from model A.

We estimate the regression models with breaks in March 1997, February 1998, September 1999, October 2000, and April 2003. Only the breaks in February 1998 and September 1999 are statistically significant. We therefore drop the other breaks from the model.

Table 1. Descriptive Evidence on the Intertemporal Allowance Market

Year	Allowance Quantities			Market-Clearing Allowance Prices in Annual EPA Auction		
	Annual Allocation (tons)	Annual Use (tons)	End-of-Year Stock (tons)	Current Vintage ($/ton)	+7 Years Vintage ($/ton)	Discount, +7 Price to Current Price (%)
1995	8,744,081	5,298,429	3,445,652	132.00	126.00	4.5
1996	8,296,548	5,433,351	6,298,986	66.05	63.01	4.6
1997	7,147,464	5,474,440	7,961,359	106.75	102.15	4.3
1998	6,969,165	5,298,498	9,630,343	115.01	108.30	5.8
1999	6,990,132	4,944,676	11,673,436	200.55	167.55	16.5
2000	9,966,531	11,201,999	10,372,487	126.00	55.27	56.1
2001	9,553,657	10,633,035	9,297,048	173.57	105.72	39.1
2002	9,542,478	10,193,684	8,648,932	160.50	68.00	57.6
2003	9,541,085	10,595,944	7,598,984	171.80	80.00	53.4
2004	9,541,085	10,259,771	6,873,273	260.00	128.00	50.8
2005	9,539,575	10,222,847	6,173,001	690.00	260.00	62.3

Notes: One allowance gives the right to emit one ton of SO_2. Allowance allocations increased substantially in 2000 at the beginning of the program's Phase II. A 60-day reconciliation period follows the end of the calendar year, so that the end-of-year stock for a given year is determined on March 1 of the following year. The annual EPA auction occurs in late March and includes sales of the current vintage and a future vintage (+7 years) of allowances. Source: U.S. Environmental Protection Agency, 2006a and 2006b.

Table 2. Summary Statistics

Variable	Units	Mean	Standard Deviation
p_t	\$/ton	146.27	38.70
Dependent variable:			
$p_{t+1} - p_t$	\$/ton	0.64	9.83
Base model:			
$r_t^f p_t$	\$/ton	0.49	0.21
$(r_t^m - r_t^f) p_t$	\$/ton	0.47	6.15
Market shocks:			
$elecusefe_{t+1}$	megawatt-hr./month	368,356	7,036,906
$lscprcfe_{t+1}$	¢/million Btu	9.56	15.03
$hscprcfe_{t+1}$	¢/million Btu	5.94	14.42
$ngasprcfe_{t+1}$	¢/million Btu	45.97	105.02
$wagefe_{t+1}$	\$/hour	0.15	0.17
Arbitrage pricing theory:			
$CPIfe_{t+1}$	*unitless*	-1.48	0.88
$10yrbondfe_{t+1}$	percentage points at monthly rates	0.0002	0.0005
$IPIfe_{t+1}$	*unitless*	2.14	4.80

Note: 112 monthly observations, 9.1994 to 12.2003.

Table 3. Explaining SO_2 Allowance Price Movements

	Base Model				Base Model and Market Shocks			
	With Breaks		Without Breaks		With Breaks		Without Breaks	
Variable	Coef.	Std. Err.	Coef.	Std. Err.	Coef.	Std. Err.	Coef.	Std. Err.
constant	3.98	(2.77) [3.84]	3.28	(2.37) [2.81]	3.34	(2.90) [2.18]	1.09	(2.84) [2.60]
Break1	8.17	(2.85)*** [3.36]**	---	---	8.21	(2.82)*** [3.25]**	---	---
Break2	-6.89	(2.90)** [3.96]*	---	---	-12.05	(3.31)*** [3.41]***	---	---
$r_t^f\, p_t$	-10.97	(4.85)** [7.49]	-5.52	(4.46) [5.50]	-13.20	(4.87)*** [4.06]***	-8.51	(4.95)* [4.54]*
$(r_t^m - r_t^f)p_t$	0.05	(0.15) [0.20]	0.09	(0.15) [0.21]	0.05	(0.15) [0.19]	0.15	(0.16) [0.21]
$elecusefe_{t+1}$	---	---	---	---	-3.07e-08	(1.45e-07) [1.38e-07]	6.78e-08	(1.50e-07) [1.60e-07]
$lscprcfe_{t+1}$	---	---	---	---	0.08	(0.08) [0.06]	0.05	(0.08) [0.08]
$hscprcfe_{t+1}$	---	---	---	---	0.01	(0.09) [0.07]	-0.04	(0.09) [0.09]
$ngasprcfe_{t+1}$	---	---	---	---	0.04	(0.01)*** [0.01]***	0.02	(0.01)** [0.01]***
$wagefe_{t+1}$	---	---	---	---	9.45	(5.84)*** [4.30]***	15.75	(5.90)*** [5.49]***
R^2	0.09		0.02		0.25		0.15	
N	112		112		112		112	

Notes. The dependent variable is the monthly change in SO_2 allowance prices, $p_{t+1} - p_t$. One, two, or three asterisks indicate significance at the levels p<0.10, p<0.05 or p<0.01, respectively. Conventional standard errors are in parentheses; Newey-West standard errors are in brackets. The variable *break1* is a dummy variable for a structural break beginning February 1998; *break2* is a dummy variable for a structural break beginning September 1999.

Table 4. Robustness Checks; Base Model and Lagged Market Shocks

	Base Model and Lagged Market Shocks			
	With Breaks		Without Breaks	
Variable	Coef.	Std. Err.	Coef.	Std. Err.
constant	5.09	(2.84)*	2.77	(2.75)
		[1.99]**		[2.34]
break1	7.55	(2.85)***	---	---
		[3.44]**		---
break2	-11.26	(3.24)***	---	---
		[3.91]***		---
$r_t^f p_t$	-14.91	(4.81)***	-10.35	(4.84)**
		[4.03]***		[5.07]**
$(r_t^m - r_t^f)p_t$	0.01	(0.15)	0.08	(0.16)
		[0.19]		[0.21]
$elecusefe_t$	-7.81e-08	(1.44e-07)	-2.51e-08	(1.50e-07)
		[1.17e-07]		[1.17e-07]
$lscprcfe_t$	-0.03	(0.08)	-0.06	(0.08)
		[0.07]		[0.08]
$hscprcfe_t$	0.04	(0.09)	-0.01	(0.08)
		[0.07]		[0.09]
$ngasprcfe_t$	0.04	(0.01)***	0.03	(0.009)***
		[0.009]***		[0.008]***
$wagefe_t$	9.79	(5.89)	16.05	(5.88)***
		[7.40]		[6.96]**
R^2	0.27		0.18	
N	112		112	

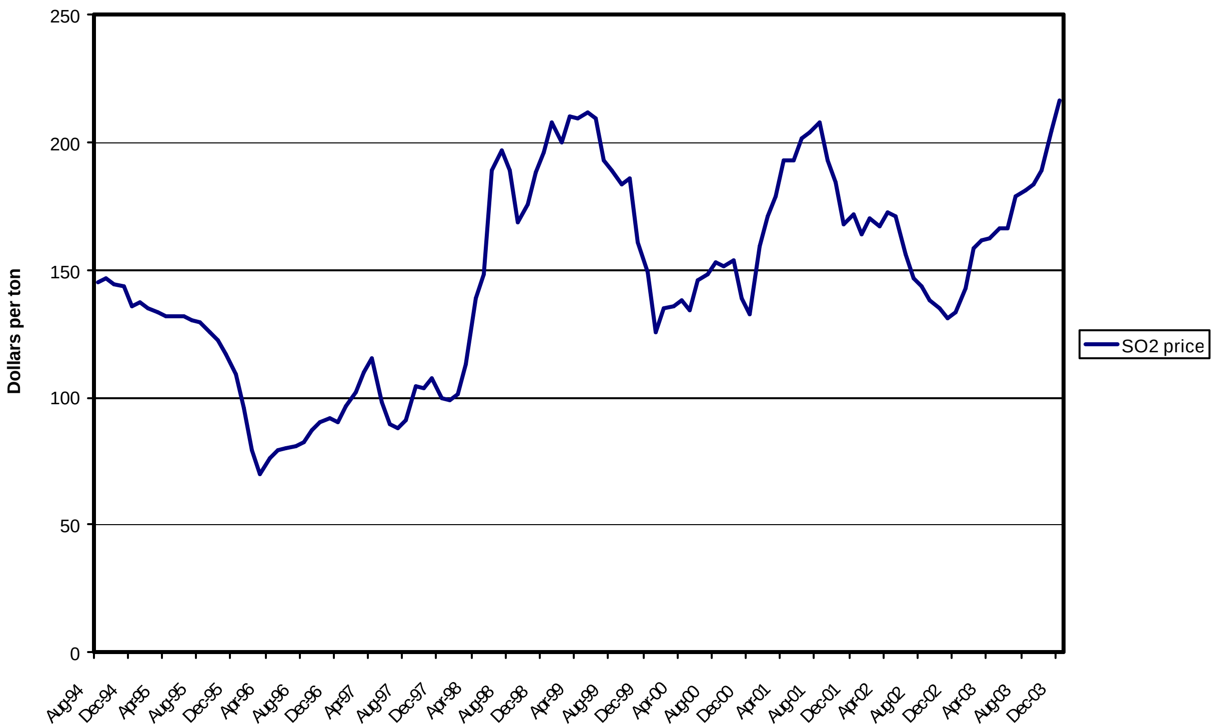
Notes. The dependent variable is the monthly change in SO_2 allowance prices, $p_{t+1} - p_t$. One, two, or three asterisks indicate significance at the levels p<0.10, p<0.05 or p<0.01, respectively. Conventional standard errors are in parentheses; Newey-West standard errors are in brackets.

Table 5. Robustness Checks; Base Model, Market Shocks, and Arbitrage Pricing Theory

	Base Model, Market Shocks, and Arbitrage Pricing Theory Variables			
	With Breaks		Without Breaks	
Variable	Coef.	Std. Err.	Coef.	Std. Err.
constant	9.79	(4.63)**	1.32	(4.28)
		[4.49]**		[4.06]
break1	14.66	(4.21)***	---	---
		[4.83]***		---
break2	-16.74	(4.62)***	---	---
		[5.00]***		---
$r_t^f p_t$	-22.47	(7.16)***	-14.43	(6.61)**
		[7.90]***		[5.60]**
$(r_t^m - r_t^f)p_t$	0.08	(0.15)	0.07	(0.16)
		[0.19]		[0.20]
$elecusefe_{t+1}$	-3.03e-08	(1.45e-07)	1.13e-08	(1.53e-07)
		[1.42e-07]		[1.43e-07]
$lscprcfe_{t+1}$	0.07	(0.08)	0.07	(0.08)
		[0.06]		[0.07]
$hscprcfe_{t+1}$	0.007	(0.09)	0.02	(0.09)
		[0.08]		[0.09]
$ngasprcfe_{t+1}$	0.03	(0.01)**	0.03	(0.01)***
		[0.01]***		[0.01]***
$wagefe_{t+1}$	7.73	(5.94)	12.67	(6.13)**
		[4.75]		[5.03]**
$CPIfe_{t+1}$	2.95	(1.87)	-0.87	(1.59)
		[1.57]*		[1.41]
$10yrbondfe_{t+1}$	2777	(2671)	-1988	(2491)
		[2786]		[3004]
$IPIfe_{t+1}$	0.35	(0.42)	0.57	(0.37)
		[0.45]		[0.33]*
R^2	0.29		0.18	
N	112		112	

Notes. The dependent variable is the monthly change in SO_2 allowance prices, $p_{t+1} - p_t$. One, two, or three asterisks indicate significance at the levels p<0.10, p<0.05 or p<0.01, respectively. Conventional standard errors are in parentheses; Newey-West standard errors are in brackets.

Figure 1. SO2 Spot Market Prices, Aug 1994 - Dec 2003



When to Pollute, When to Abate? Evidence on Intertemporal Permit Use in the Los Angeles NO_x Market

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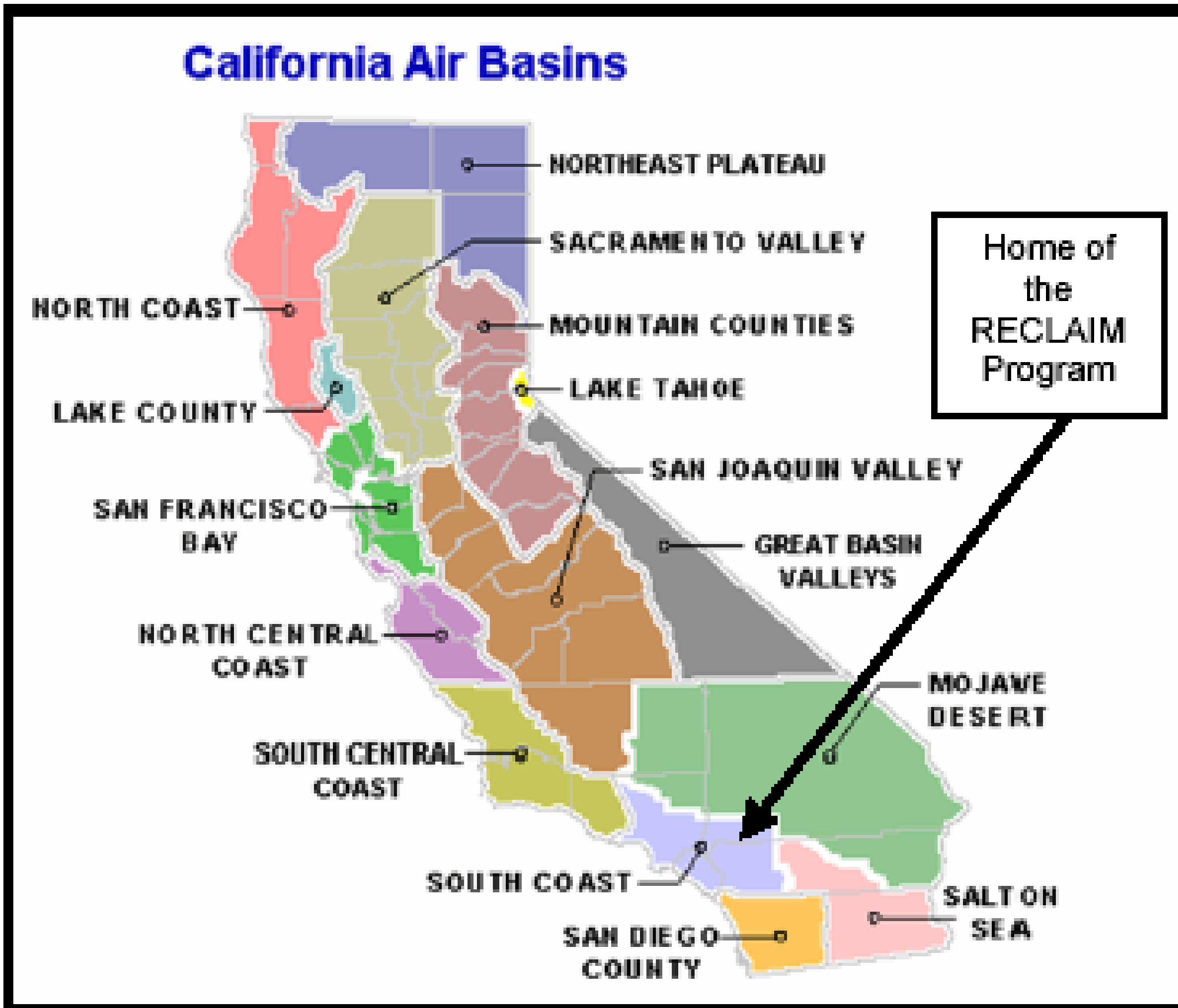
Motivation for Study

- Tradable permit programs
  - Numerous programs; many proposed
  - Politically feasible; cost effective
- Market design issues
  - Bankable??
    - Short cycle: less intertemporal trading
    - Long cycle: potential hotspots or non-attainment
- RECLAIM overlapping permit cycles
  - Equilibrium properties: cost effective?
  - Empirical analysis: consistent w/ equilibrium?

The RECLAIM Program

- RECLAIM: Regional Clean Air Incentives Market
- Implemented January 1994
- Goal: Compliance with National Ambient Air Quality Standards (NAAQS) by 2003
- NO_x and SO_2 caps declining annually
  - 75% decrease in NO_x cap, 1994-2003
- Stationary sources
- Heterogeneous industries (“facilities”)
  - Electricity generators, petroleum refineries, cement factories, many others
- Four counties in the Los Angeles smog airshed

South Coast Air Quality Management District (SCAQMD)

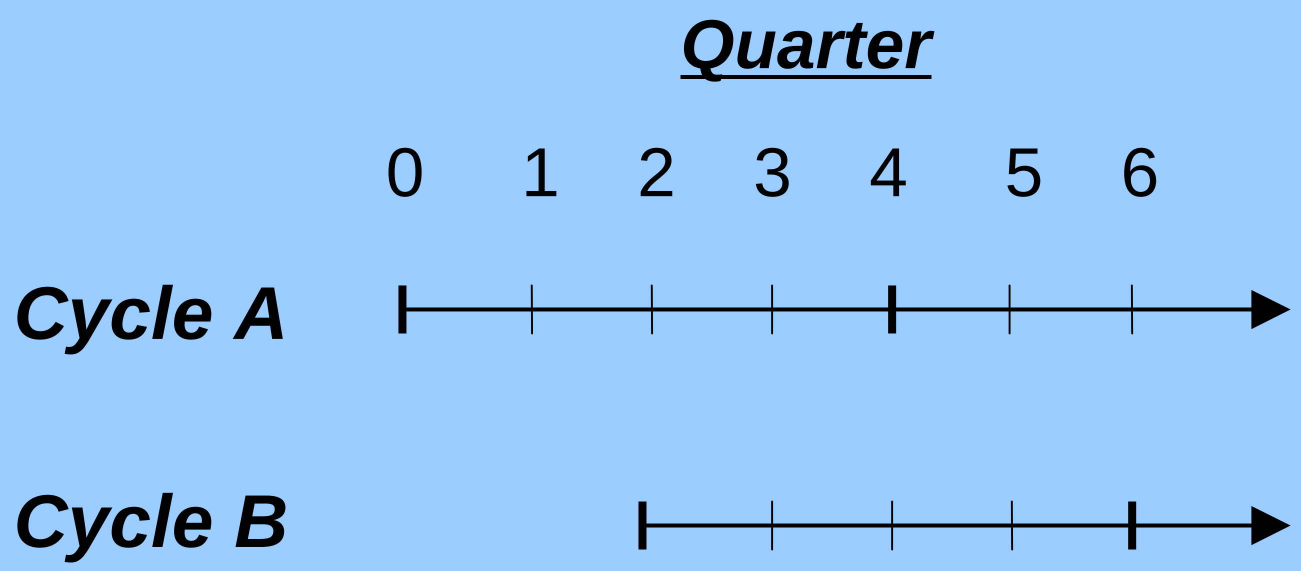


The RECLAIM Market

- RECLAIM trading credit, or RTC
  - 1 pound of emissions
- Tradable, but not bankable
- Annual compliance

- Overlapping compliance cycles
  - Cycle A: January-December compliance year
  - Cycle B: July-June compliance year
- Facilities are assigned to a cycle
- Permits are tradable between cycles
  - Cycle A facility can buy and use a Cycle B permit

Overlapping Compliance Cycles



-- Emissions reported quarterly --

-- Annual compliance --

Modeling the RECLAIM Market

- Dynamic model
  - Time as quarters (quarterly reporting of emissions)
- Supply side
  - Semi-annual permit caps
  - E_t = supply of permits that expire in quarter t
  - $E_t > 0$ if t is even; $E_t = 0$ if t is odd
- Demand side
  - Firm objective function: minimize the discounted sum of abatement costs and permit costs

- Demand side (continued)

$$\min_{d_t^A d_t^B} \sum_{t=1}^{\infty} \delta^t c_t(a_t) + \delta^{t+i} p_{t+i}^{t+i} d_t^A + \delta^{t+i} p_{t+i}^{t+i+j} d_t^B$$

where

a_t = abatement

$c(a_t)$ = abatement cost function

d_t^A = demand for cycle A permits

d_t^B = demand for cycle B permits

p_{t+i}^{t+i} = price of cycle A permits at compliance time

p_{t+i}^{t+i+j} = price of cycle B permits at compliance time

δ = discount factor

$a_t = \varepsilon_t - d_t^A - d_t^B$, with ε_t = counterfactual emissions

- Demand side (continued)
  - Demand correspondences for permits of each cycle for each quarter
  - Defined for firms in compliance cycle A and for firms in compliance cycle B
  - Aggregate demand sums the individual demands

- Equilibrium
  - Intertemporal arbitrage: $p_t^\tau = p_0^\tau \delta^{-t} = p_0^\tau (1+r)^t$
  - Equilibrium condition:

$$\delta^t c_t'(a_t) = \delta^{t+i} \min\{p_{t+i}^{t+i}, p_{t+i}^{t+i+j}\} = \min\{p_0^{t+i}, p_0^{t+i+j}\}$$

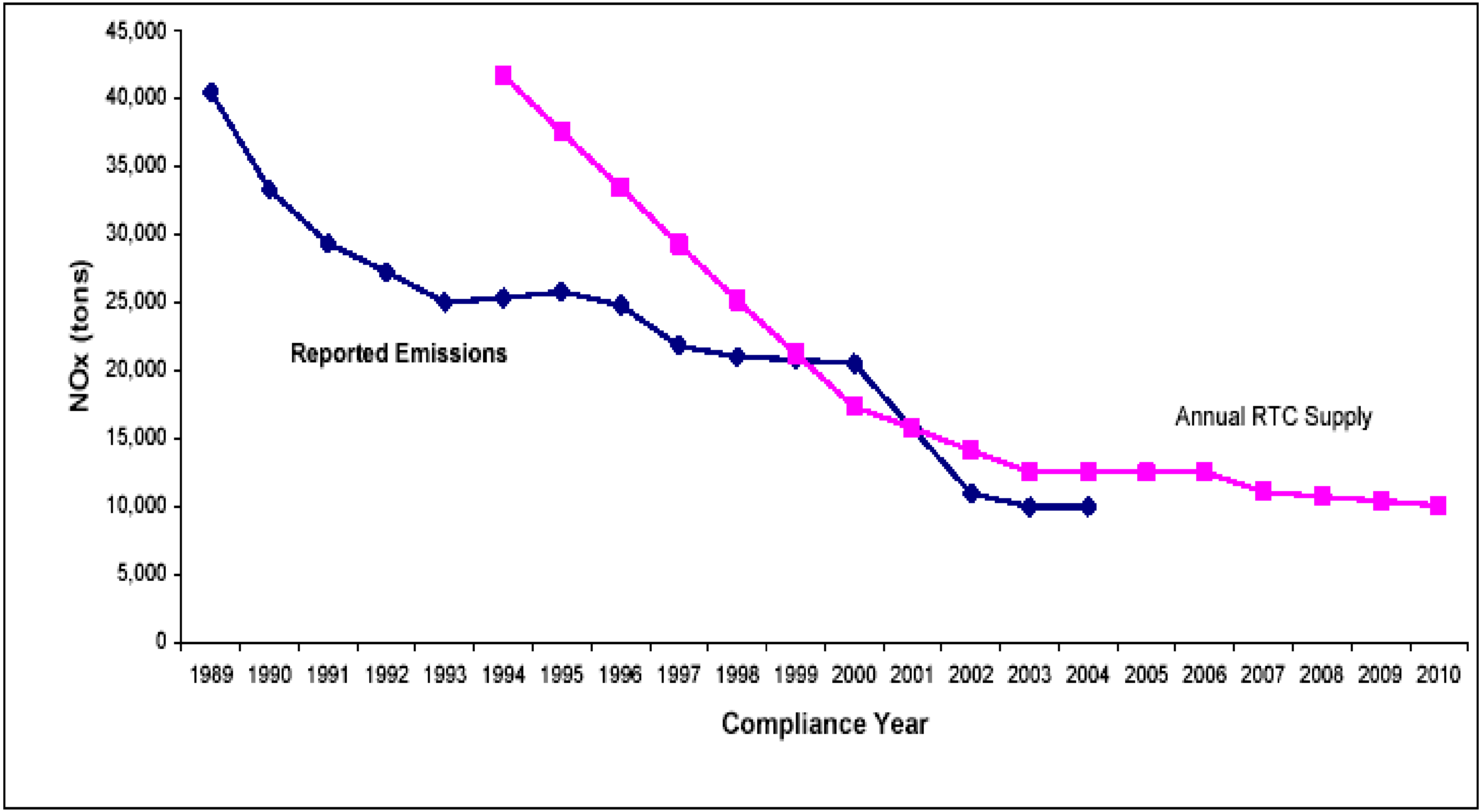
Main Theoretical Results

- A competitive equilibrium exists
  - It is cost effective
  - It is not necessarily dynamically efficient
- The equilibrium is invariant to:
  - Reassigning a firm from Cycle A or Cycle B to the other cycle
  - Reallocating the initial endowment of permits
- Emissions are higher in quarter *t-1* than in quarter *t* (where *t* is a compliance quarter)
  - Qualifying conditions: positive prices and controlling for abatement costs

Data and Variables

- Panel data on emissions
  - Facility-level (cycles A and B)
  - Quarterly, 1994-2003
  - NO_x and SO_2
  - > 400 facilities and > 12,000 observations (NO_x)
- Control variables
  - Fixed effects
  - SIC codes
  - Annual endowment of permits
  - Producer prices
  - Actual and average temperatures
  - Zone (coastal or not)

Two Phases:
Nonbinding and Binding Caps



Proposition: *Emissions are higher in quarter t-1 than in quarter t (where quarter t is a compliance quarter)*.

- Difference-in-differences estimator:

$$e_{it} = \alpha + \beta_1 CmplncQtr + \beta_2 Scrcty + \beta_3 CmplncQtr * Scrcty + \nu_i + \varepsilon_{it}$$

where

e_{it} = NO_x emissions by facility *i* in quarter *t*

CmplncQtr = dummy variable for last quarter in cycle

Scrcty = dummy variable for scarcity phase

ν_i = facility fixed effects

ε_{it} = error term

$$e_{it} = \alpha + \beta_1 CmplncQtr + \beta_2 Scrcty + \beta_3 CmplncQtr * Scrcty + \nu_i + \varepsilon_{it}$$

β_1 = Average difference in quarterly emissions between quarters *t-1* and *t* in pre-scarcity phase

β_2 = Average change in quarter *t-1* emissions after entering scarcity phase

β_3 = Average difference between quarter *t* and quarter *t-1* changes in emissions after entering scarcity phase

>> <u>Hypothesis</u>: $\beta_3 < 0$

Delayed Abatement

Difference in Differences

Dependent variable: quarterly NOx emissions

	Spec. 1	Spec. 2	Spec. 3
Coeff	1686	1504	5162*
Std Err	1798	1789	1854
CmplncQtr	Yes	Yes	Yes
Scrcty	Yes	N.A.	Yes
Year Dummies	No	Yes	No
Facility F.E.	Yes	Yes	Yes
N	14,089	14,089	11,687
Facilities	530	530	528

Note: Model predicts negative coefficient.

Proposition: *Assignment of a firm to Cycle A or Cycle B does not affect quarterly emissions*.

DID Estimator:

$$e_{it} = \alpha + \beta_1 LateQtr + \beta_2 Scrcty + \beta_3 LateQtr * Scrcty + \nu_i + \varepsilon_{it}$$

where

e_{it} = NO_x emissions by facility *i* in quarter *t*

LateQtr = d.v. for last two quarters of compliance year

Scrcty = d.v. for scarcity phase

ν_i = facility fixed effects

ε_{it} = error term

>> <u>Hypothesis</u>: $\beta_3 = 0$

Predictive Power of Cycles

Difference in Differences
Dependent variable: quarterly NOx emissions

	Spec. 1	Spec. 2	Spec. 3
Coeff	-1555	-2686	-2684
Std Err	2088	2095	1823
LateQtr	Yes	Yes	Yes
Scarcity	Yes	N.A.	Yes
Year Dummies	No	Yes	No
Facility F.E.	Yes	Yes	Yes
N	12,014	12,014	10,125
Facilities	403	403	403

Note: Model predicts zero coefficient.

Summary

- Market design issues moving to forefront
- RECLAIM's overlapping cycles feature
  - Limited intertemporal trading
  - Cost effective
  - Reasonable for some pollutants and certain regulatory contexts
- Tests of theoretical propositions underway

Preliminary Draft
Do Not Quote
Comments Welcome
Last updated on 10/3/06

A Spatial Analysis of the Consequences of the SO2 Trading Program

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Abstract

Title IV of the 1990 Clean Air Act Amendments (CAAA) set a cap on the SO_2 emissions of the dirtiest coal-fired electric utilities at 9 million tons per year (roughly 50% below their 1980 levels, to be fully implemented in 2010). At the same time, Title IV significantly changed the manner in which coal-fired utilities were regulated from command-and-control emission standards to a system of tradable allowances. In this paper we examine the level of the health benefits and abatement costs associated with the air quality improvements mandated under Title IV and compare them with the level of health benefits and abatement costs that might have occurred from a comparable reduction in emissions using a command-and-control system. Using data for 148 coal-fired utilities during the first year of Title IV (1995), we find as expected that the benefits of reduced SO_2 emissions under Title IV greatly exceeded the costs: we estimate benefits of nearly $56 billion and costs of only $558 million. We then compare the health benefits and abatement costs under allowance trading versus a hypothetical command-and-control system requiring the same overall level of emission reductions. We find that the allowance trading system led to sizable savings (16.8%) in abatement costs, but that allowance buyers tended to have emissions with higher marginal benefits (damages) than sellers, more than offsetting the savings in abatement costs. This result suggests a possible role for spatially-based 'exchange rates' in allowance trading. We explore the possibility of spatially-based allowance systems, such as trading regions, but find that considerable heterogeneity in marginal benefits within regions limits the potential gains from such systems.

I. Introduction

During the late 1980's, prior to the passage of Title IV of the 1990 Clean Air Act Amendments (CAAA), there had been a spirited debate involving Congress, the Environmental Protection Agency (EPA), and academics, about the importance of reducing sulfur dioxide (SO_2) emissions due to the problem of acid rain. Acid rain occurs when SO_2, released as a gas from coal when it is burned at high temperatures, reacts with water in the atmosphere to form sulfurous acid and sulfuric acid and then returns to earth in the form of raindrops and dry particles. Some of the acid rain caused by SO_2 emissions from coal-fired utilities in the upper Midwest falls in Canada. Thus, in addition to domestic pressure to reduce SO_2 emissions, Canada was also putting political pressure on the U.S. to decrease its SO_2 emissions. Soon after the passage of the CAAA the U.S. and Canada formally agreed to control transboundary acid rain by signing the Canada-United States Air Quality Agreement.

The ecological damage from acid rain, while important, is relatively minor when compared to decreases in premature mortality from SO_2 reduction. For example, Burtraw et al (1997) estimate the expected environmental benefits from recreational activities, residential visibility, and morbidity from the Acid Rain Program to be only $13 per capita in 1990. On the other hand, in 2002 the EPA estimated that, by 2010, human health benefits from the Acid Rain Program will be approximately $50 billion annually (due to many fewer cases of premature mortality, fewer hospital admissions and fewer emergency room visits). These human health benefits mainly arise from lower ambient levels of secondary particles (PM_{10} and $PM_{2.5}$) – which have been linked in numerous studies to premature mortality – which form when SO_2 combines with ammonia in the atmosphere.

Most of the SO_2 emissions in the United States come from coal fired electric utilities. Title IV of the 1990 CAAA establishes an annual emissions cap of 9 million tons of SO_2

emissions from all fossil-fuel fired electric utilities over 25 megawatts, to be fully implemented by 2010. This annual cap requires the affected electric utilities to reduce their total SO_2 emissions by 10 million tons below their 1980 levels. Title IV also significantly changed the manner in which coal-fired utilities were regulated from command-and-control emission standards to a more flexible, cost-efficient system of allowance trading. The more flexible allowance trading approach made the considerable SO_2 reductions politically feasible and is generally thought to have led to large cost savings relative to the previous command-and-control approach. For example, Keohane (2003) estimated that the allowance trading system resulted in annual cost savings between \$150 million and \$270 million relative to a uniform emissions-rate standard. Furthermore, the tremendous flexibility of the allowance trading program provides the market with the proper incentives to produce an efficient allocation of SO_2 reductions, if SO_2 emissions have the same marginal benefit everywhere across the United States. However, our estimates of the health benefits resulting from SO_2 reductions indicate substantial heterogeneity across plants in the marginal benefit per ton of SO_2 reduced. Therefore, since Title IV allows one-to-one allowance trading, we should not expect the resulting allocation of emission reductions to maximize the net benefits from SO_2 reductions.

In this paper we extend the work of Shadbegian, Gray, and Morgan (2006) by examining two different scenarios of SO_2 reductions leading to significant air quality improvements. In one scenario, we measure these improvements relative to the level of emissions under the former command-and-control regime, which allowed a greater level of emissions. In another scenario, we measure the improvements relative to a counterfactual distribution of emissions based on requiring emissions reductions similar in magnitude to those actually achieved under Title IV, but imposed on plants through a reduction in the allowable emissions rate for all plants, without the possibility of trading.

The overwhelming majority of the dollar-valued benefits from air quality improvements come from the impact of airborne fine particulate matter ($PM_{2.5}$ and PM_{10}) on premature mortality. In 1997 the EPA estimated that \$20 trillion dollars of the estimated \$22.2 trillion dollars worth of benefits derived from the Clean Air Act of 1970 (between 1970 and 1990) resulted from reductions in particulate-related premature mortality. In this paper, we use a spatially-detailed air pollution receptor model (the Source-Receptor Matrix) to model the impact that SO_2 emissions have on $PM_{2.5}$ concentration levels in each county in the United States during 1995, the first year of Title IV. We then use information from the epidemiology literature on the correlation between exposure to $PM_{2.5}$ and mortality to translate the reductions in secondary $PM_{2.5}$ concentrations in each county in the U.S. into the dollar benefits from reductions in premature mortality.

Are the substantial air quality improvements due to lower SO_2 emissions costless? The answer could be yes if increases in efficiency resulting from the new allowance trading system (e.g. more flexibility in complying with regulations, less uncertainty about future regulatory requirements) more than offset the extra abatement costs on a plant-by-plant basis. However, a more likely outcome is that some plants will still face higher abatement costs, which will be passed along to their customers. Furthermore, if some plants buy SO_2 allowances to increase their emissions (or at least not to lower them by as much as they otherwise would have), the population impacted by the worsening air quality (or at least the relatively less clean air) will be 'paying' some of the costs of the greater air quality improvements near other plants that reduced their emissions in order to sell SO_2 allowances. In addition to comparing the costs and benefits that arise from lower SO_2 emissions under Title IV, we simulate the impact of requiring a comparable reduction in overall SO_2 emissions under the old command-and-control regime, assuming that a uniform emission standard is in place at all plants.

Using data for the 148 dirtiest coal-fired utilities we find, as expected, that the aggregate benefits in 1995 from lower SO_2 emissions under Title IV greatly exceed their costs: we estimate benefits of $56 billion (a bit larger than EPA's estimates of total benefits of $50 billion by 2010) and costs of only $558 million. Therefore, the net benefits from the SO_2 reduction are roughly $55 billion or $100 in benefits for every $1 in abatement costs. Comparing the consequences of requiring similar overall emissions reductions using command-and-control regulation, we find that trading results in significantly lower costs ($94 million or 16.8% lower). However, shifts in the spatial distribution of emissions tend to lower aggregate benefits from SO_2 reductions, since allowance buyers have emissions with higher marginal benefits (damage) than allowance sellers. This result suggests the possibility of limiting trades between plants, either by defining trading zones that would allow only trades between plants in the same zone, or by developing some sort of 'exchange rate' for allowance trades, based on the relative marginal benefits of the two plants involved. We explore the possibility of trading zones, but find that considerable heterogeneity in marginal benefits within regions limits the potential gains from such systems.

The rest of the paper is organized as follows. In section II we present background information on Title IV of the CAAA of 1990. Section III contains a brief survey of the literature on studies examining various aspects of the Title IV trading program. Section IV describes the methodology we use to estimate both the health benefits and the costs of SO_2 abatement under Title IV and Section V describes our sample of plants. In Section VI we discuss our findings and we end with some concluding remarks in Section VII.

II. Title IV: Background Information

Title IV of the 1990 CAAA significantly changed the manner in which coal-fired utilities were regulated in the U.S. Before Title IV utilities were regulated by command-and-control emission standards, where utilities were required to meet individual emission standards set by regulators. Title IV established a more flexible, cost-efficient cap-and-trade program that set a cap on total SO_2 emissions, allocated allowances among generating units equal to that cap, and allowed plants to freely trade these allowances among their own units, to sell them to other plants, or to bank them for future use.[1] The only requirement imposed on a plant under the allowance trading program is that, at the end of the year, it must have one allowance for each ton of SO_2 emitted that year. Thus, the allowance trading program created by Title IV provides more flexibility to comply with any given emission standard, because utilities which have high marginal abatement cost may purchase SO_2 allowances from utilities which have lower marginal abatement costs.

The overall goal of Title IV was to decrease total SO_2 emissions to roughly 9 million tons by 2010, approximately half of the 1980 level. The reduction was to be accomplished in two

[1] The only time a plant is denied the right to buy allowances is when that plant is located in a county which is in violation of the National Ambient Air Quality Standard (NAAQS) for SO_2, which is set at a level to prevent local adverse health outcomes. However, this has not proved to be a major hindrance in the SO_2 allowance market since the Title IV cap requires a considerably larger reduction of aggregate SO_2 emissions than what is required to meet the NAAQS for SO_2.

phases. Phase I, which occurred from 1995-1999 targeted the dirtiest 110 power plants with 263 generating units. These generating units, referred to as the Table A units, were required to lower their aggregate emissions to 7.2 million tons per year in 1995, 6.9 million tons in 1996, and then 5.8 million tons from 1997-1999. In 1990, together the Table A units emitted 8.7 million tons of SO_2, but they only emitted 4.5 million tons in 1995 (nearly 50% less). During Phase I the initial number of allowances a generating unit was allocated was determined by multiplying its average 1985-1987 heat input by an average emission rate of 2.5 lbs of SO_2 per million BTUs of heat input.[2] Each SO_2 allowance gave a generating unit the right to emit one ton of SO_2, and at the end of the year the generating unit could only emit an amount of SO_2 equal to the number of allowances it held.[3]

Phase II, which began in 2000, expanded the cap-and-trade program to include any fossil-fueled fired generating units with an output capacity of 25 megawatts or greater.[4] In addition to including most of the smaller and cleaner units, Phase II also required the Table A units to make further reductions in their SO_2 emissions – reducing their aggregate SO_2 emissions by an additional 3.4 million tons, down to 2.4 million tons by 2010. During Phase II basic annual allowance allocations to each generating unit are based on an average emission rate of 1.2 lbs of SO_2 per million BTUs of heat input, a much more stringent standard than the emission rate of 2.5 lbs during Phase I.

Two additional provisions of Title IV – 'substitution' and 'compensation' – allow other generating units not required to make reductions during Phase I to voluntarily come under Title IV along with the Table A units. The substitution provision allows Table A units to contract for emission reductions at non-Table A units instead, thereby reducing the cost of SO_2 reduction. On the other hand, the compensation provision prevents Table A units from meeting their emission reductions by simply reducing generation. In other words, if a Table A unit significantly reduces its generation below its baseline levels then it must bring one or more non-Table A units under Phase I regulation to compensate. The increased generation at the non-Table A units must offset the reduction at the Table A unit.

The total number of allowances available to participating units in 1995 was 8.7 million. The initial allocation of allowances issued to the Table A units was approximately 5.55 million. The number each unit received was based on their historical coal use and emission rates. The 'compensating' and 'substitution' units were granted a total of 1.33 allowances. Additional allowances were also issued through allowance auctions (175,000 in 1995) and through other bonus provisions in the CAAA including: Phase I Extension Allowances; Early Reduction Credits; Small Diesel Allowances; and Conservation Allowances. A total of 1.35 million Phase I Extension Allowances were allocated to Phase I units that either reduce their emissions by 90% or transferred their reductions to other units that reduce their emissions by 90%. Approximately 314,000 Early Reduction Credits were allocated to units that voluntarily reduced their emissions between 1990 and 1995. Slightly more than 50,000 allowances were issued as conservation and small diesel allowances. Small diesel allowances were given to small diesel refineries in 1995 that manufactured and desulfurized diesel fuel in 1994, while conservation allowances were earned by plants that undertake efficiency and renewable energy measures.

[2] Note allowances are allocated to individual generating units and not to plants.
[3] Generating units face a fine of $2000 for each ton of SO_2 emitted for which they do not have an allowance.
[4] Some of these smaller generating units (111) joined Phase I, under the "substitution" and "compensation" provisions of the CAAA, and are included in this analysis.

During 1995 SO_2 emissions from Phase I generating units dropped significantly.[5] Phase I plants emitted only 4.9 million tons of SO_2, 4.6 million tons less than they emitted in 1990 – 3.2 million tons less than was required by Title IV. However, large decreases in SO_2 emissions were observed just after the passage of Title IV, even before the trading system was in place and plants were required to make large reductions. There have been several explanations offered to help explain the pre-1995 reductions. First, plants may have acted strategically by complying early with Title IV. Early compliance would allow utilities to pass on to consumers the additional higher cost of low-sulfur coal and/or the cost of installing scrubbers. Second, certain states revised their State Implementation Plans requiring electric utilities to lower their SO_2 emissions prior to 1995. However, the most probable explanation is that the deregulation of railroads made it much less expensive to ship low-sulfur coal from the Powder River Basin to Midwest, the geographic region which experienced the greatest SO_2 reductions between 1985 and 1993 (Ellerman and Montero, 1998).

Finally, the SO_2 cap-and-trade program builds in even more flexibility by letting allowances that are not used in one year to be 'banked' and used in any later year. In other words, a plant can lower its emissions below their annual allowance allocation, thereby not exhausting their allotment of allowance and 'deposit' the extra allowances in an 'emissions bank.' These 'banked' allowances are perfect substitutes for future year allowances, and may be used or sold. Phase I plants 'banked' many allowances from 1995-1999 most likely to smooth the transition the more stringent limits imposed under Phase II starting in 2000. In particular, plants banked more than 11.5 million allowances during Phase I (1995-1999). Plants then used 1.2 million of these banked allowances in 2000, the first year of Phase II, followed by 1.08 million allowances in 2001 and another 650,000 million allowances in 2002. This systematic drawing down of the allowance bank suggests that the over compliance during Phase I was intentional (rather than being an unexpected result of lower than expected prices for low-sulfur coal).

III. SO_2 Trading Program: Literature Review

Prior to the introduction of emissions trading, Gollop and Roberts (1985) showed that a cost-effective allocation of pollution abatement arising from allowance trading among electrical utilities could produce an almost 50% reduction in abatement costs, suggesting potentially huge savings from emissions trading. In the years since the advent of Title IV, many papers, including Burtraw et al (1997), Joskow et al (1998), Schmalensee et al (1998), Carlson et al (2000), Popp (2000), Keohane (2002,2003), Ellerman (2003), and Shadbegian and Morgan (2003), have examined many different aspects of the actual SO_2 allowance trading program including its cost savings, environmental effectiveness, spatial patterns of abatement, pollution control innovations, and the efficiency of the banking of allowances. The likely success of any pollution allowance-trading program depends critically on the efficiency of the allowance trading market. Joskow et al (1998) evaluate the efficiency of the SO_2 allowance market by comparing the price of allowances auctioned by EPA between 1993 and 1997 with private market allowance price indices. If the SO_2 allowance market is efficient then EPA auction prices and private market prices will be equal. Joskow et al find that by the end of 1994 EPA auction prices and private

[5] Recall our analysis is done at the plant level, but regulation of the electric utilities takes place at the generating level. Phase I plants include the 110 plants (with 263 generating units) that were regulated under Phase I plus the 38 plants (111 generating units) that opted into Phase I.

market prices for SO_2 allowances were virtually identical implying that the private market for tradable allowances was relatively efficient. Furthermore, Schmalensee et al (1998) also conclude that the private market for tradable allowances was relatively efficient by noting the tremendous growth in the number of market trades from 1995 to 1997: 1.6 million, 4.9 million, and 5.1 million allowances were traded, respectively.

Keohane (2003) concludes that Title IV's allowance trading system resulted in annual cost savings between $150 million and $270 million relative to a command-and-control uniform emissions-rate standard. On the other hand, Carlson et al. (2000) find that the sizeable decrease in pollution abatement costs during the beginning of Title IV relative to the initial estimates was due more to the technological progress that lowered the cost to switch to low sulfur coal and the reduction in the price of low sulfur coal rather than the ability to trade allowances per se. Shadbegian and Morgan (2003) examine the impact of the stringency of SO_2 regulations on the productivity of electric utilities before and after the implementation of Title IV. They estimate that a 10% increase in regulatory stringency lowered productivity by 0.66% prior to Title IV, while during Title IV that same increase in regulatory stringency had no significant impact on productivity. The productivity gain is equivalent to 31 million more kilowatts (kwh) of electricity – equivalent to $1.5 million cost savings, evaluated at $0.05/kwh.

Ellerman (2003), among other issues, examines whether or not the more than 11 million allowances 'banked' during Phase I was optimal. He concludes that, given a reasonable set of assumptions concerning both the discount rate and the expected growth of SO_2 emissions during the banking period, the level of banking that took place during Phase I was consistent with rational, cost-minimizing behavior on the part of the electric utilities.

Beyond the direct cost-savings that arise from the use of market-based mechanisms to protect the environment, economists have argued for their use because of the potential gains from induced technological change. Popp (2003) and Keohane (2002) have both provided empirical evidence that Title IV led to induced technological change. Popp shows that prior to the passage of the 1990 CAAA, regulation which mandated the use of scrubbers with a 90% removal efficiency rate in many new plants, created incentives which led to innovations that decreased the cost of operating scrubbers, yet did little to increase the ability of scrubbers to abate pollution. However, Popp provides evidence that since Title IV there has been technological innovations that have improved the removal efficiency of scrubbers. Keohane examines the choice of electric utilities' to install a scrubber or switch to low sulfur coal under command-and-control versus a more flexible system of allowance trading. He provides evidence that fossil-fuel fired electric utilities that were subject to Title IV were, for a given increase in the cost of switching to low sulfur coal, more likely to install a scrubber.

One potential reason why an allowance trading system may not maximize net benefits from emission reductions is that emissions from different sources may have different impacts on human health (or other benefits). Baumol and Oates (1988, Chapter 12) argue that differences in health impacts across different emission sources can lead to a suboptimal outcome when high marginal damage sources buy allowances from low marginal damage sources on a one-for-one basis. Tietenberg (1995) reviews the literature on the spatial effects associated with tradable allowances, arguing that the first-best option – potentially each source paying a different price for an allowance – significantly complicates the trading process, so a range of second-best options have been proposed. One second best option that has been proposed in the literature is to minimize the distortion which may arise from heterogeneous marginal damages across sources by dividing the control area into different zones. The zones should be defined such that emission sources are similar enough within a zone to allow unrestricted trading. On the other hand, trading will be permitted between zones only at a predefined trading ratio ('exchange rate') that is based on the relative marginal damages. Creating a system of trading zones is appealing since it should increase the level of net benefits relative to a completely unrestricted trading system. However, as Atkinson and Tietenburg (1982) point out, a system of trading zones has three undesirable effects: 1) it increases compliance costs by reducing the number of cost minimizing trades; 2) it makes the final allocation of air quality improvements more reliant on the initial allocation of allowances, since that allocation determines the overall level of emissions in each zone; and 3) it decreases the number of market participants which increases the likelihood of noncompetitive behavior. Furthermore, a system of trading zones places more burden on the regulator since the regulator would need to know the marginal damage function of all sources to set the optimal trading ratios ('exchange rates').

IV. The Benefits and Costs of Cleaner Air

A. Benefits from Cleaner Air

We estimate the human health benefits from SO_2 reductions (SO2BEN) from a given emission source by the change in mortality risk from exposure to ambient particulate concentrations caused by those SO_2 emissions. These human health benefits are calculated using a simplified linear damage function, based on estimated parameters from the literature:

SO2BEN = SO2DIFF*AIR_QUAL_TC * HEALTH_CHG * POP * VSL.

AIR_QUAL_TC is the transfer coefficient – the change in air quality (ambient particulate matter – $PM_{2.5}$) per ton change in SO_2 emissions (SO2DIFF). HEALTH_CHG is the change in mortality risk to the impacted population corresponding to the changes in air quality. POP is the size of the impacted population, and VSL (value of statistical life) is the dollar value associated with reducing premature mortality.

We calculate air quality changes at any given location using the Source-Receptor (S-R) Matrix Model, as described in Latimer (1996) and Abt (2000). The S-R Matrix model was initially calculated using the Climatological Regional Dispersion Model (CRDM). The model includes data on air pollution emissions from 5,905 separate sources in the U.S., along with additional sources from Mexico and Canada.[6] The S-R Matrix relates emissions of each particular pollutant from each source to the resulting ambient concentrations of each pollutant in every county in the U.S. More specifically, the S-R Matrix provides the necessary transfer coefficients to calculate the county-by-county changes in annual average pollutant concentrations for a one unit change of emissions for a particular pollutant from each source. The S-R Matrix transfer coefficients are a complicated function of numerous factors including wet and dry deposition of gases and particles, chemical conversion of SO_2 and nitrogen oxide (NO_X) into secondary particulates, effective stack height, and several atmospheric variables (including wind

[6] Emissions sources in the U.S. include ground-level sources, county-level sources and individual sources. Emissions from ground-level sources are estimated for each of the 3,080 contiguous counties (excludes Alaska and Hawaii, whereas elevated sources are grouped according to effective stack height. Point sources with an effective stack height taller than 500 meters are modeled as individual sources of emissions. All emission sources in the same county with an effective stack height less than 250 meters are aggregated into a single county-level source – the same is done for emission sources with an effective stack height between 250 meters and 500 meters. Ground-level emission sources are also aggregated to the county level. The S-R matrix models 5,905 U.S. emission sources.

speed and direction, stability, and mixing heights). We use the AIR_QUAL_TC to measure the impact of SO_2 emissions on ambient concentration of $PM_{2.5}$ in each county.

Our study concentrates on the human health benefits from lower ambient concentrations of secondary particulates ($PM_{2.5}$) that result from reductions in SO_2 emissions. We use the results from the American Cancer Society (ACS) study, the most complete analysis of long-term mortality effects from air pollution to date (Pope et al., 2002) to measure HEALTH_CHG. Pope et al. find that a 10 $\mu g/m^3$ increase in $PM_{2.5}$ concentrations leads to an approximate 4% (95% confidence interval: 2%, 6%) higher mortality rate in the exposed population. We assume that the secondary particulates formed from SO_2 have the same impact on premature mortality (Pope et al. found similar numbers for sulfate particles in their study).[7] We estimate the exposed population, POP, based on county-level data from the 1990 Census of Population, which provides the number of people living in each county (and thus the number of exposed people by the average ambient pollution concentrations in that county).

Finally, we use a recent EPA (1997) benefit-cost analysis that estimated the value of a statistical life (VSL) to put a dollar value of premature mortality. The EPA study combined contingent valuation and wage-risk studies to provide a central VSL estimate of $5.4 million (in 1995 dollars) per life saved. Note that our study assumes constant values for the VSL and HEALTH_CHG terms for the entire population. In other words, each exposed person is assigned the same average dollar harm from exposures to fine particulates and the same level of sensitivity to fine particulates.[8] Note also that the very large estimates we find for the benefits of lowering SO_2 emissions are a combination of these two factors: one will get smaller benefits by assuming either smaller health effects or a lower VSL.

B. Costs of Cleaner Air

There are three basic options (or combinations of options) available to plants to comply with Title IV: install a scrubber, switch to lower sulfur coal, or buy allowances. We measure the cost of abating a ton of SO_2 emissions in two ways. Our first estimate of the cost of complying with Title IV (COST1) is based on the actual method each plant chose to use, given the option of purchasing allowances. From Ellerman et al (1997) we have an estimate of the average cost of SO_2 abatement for each of the 374 units (plant-boiler observations) regulated by Title IV during Phase I – this consists of the 263 units mandated to reduce their SO_2 emissions by Title IV plus the 111 units which ‘opted’ into Phase I. According to Ellerman et al (1997) the average cost of ‘switching’ and ‘scrubbing’ in 1995 was $153 and $265 per ton respectively, whereas the average price of an allowance was $128.50.[9] Our second estimate of the cost of complying with Title IV (COST2) is based on Keohane (2003), which models each unit’s abatement costs based on its decision to install a scrubber or not. The decision to install a scrubber is first evaluated given the Title IV allowance trading program and then given a traditional command-and-control regime (a no trading scenario) designed to produce the equivalent aggregate SO_2 emission reductions realized under the 1990 CAAA. Keohane estimates the emissions and SO_2 abatement costs at each of the plants assuming both an emissions trading regime and a command-and-

[7] Chay and Greenstone (2003a, 2003b) analyze the impact of the exposure of fine particulate matter on infant mortality, and find similar results to the ACS study, measured in terms of increased mortality rates.

[8] Our data would readily allow our calculations to vary both in terms of sensitivity and valuation for different subpopulations – if one could generate a consensus on how to quantify such differences, a politically charged issue that we avoid here.

[9] We would like to thank Denny Ellerman for providing us with this data.

control regime, and the difference in costs between the two regimes gives us our second measure of SO_2 abatement costs. [10]

Who pays these extra abatement costs? One possible answer is “nobody”, if efficiency improvements resulting from the new allowance trading system (e.g. more flexible production switching, less uncertainty about regulatory requirements) outweighed the additional abatement costs on a plant-by-plant basis. However, a more likely scenario is that plants facing higher costs of pollution abatement will pass along these costs to their customers. We assume that all of the extra costs are passed through to the utility’s customers, and that all customers live in the same state where the utility is located.[11] We use data from the 1990 Census of Population to allocate each plant’s extra abatement costs equally to all people living within that state.

V. Sample Coverage

Phase I of Title IV regulated the emissions of 263 generating units (the Table A generating units) owned by 110 plants. An additional 38 substitution and compensation plants (111 generating units) opted into Phase I, bringing the final total to 374 generating units. Our sample consists of all 148 plants and their 374 generating units. The geographic distribution of these plants – heavily concentrated in the Midwest - is shown in Figure 1.

In Table 1 we present information on SO_2 emissions and the allocation of SO_2 allowances obtained from the EPA’s Allowance Tracking System (ATS).[12] The 148 plants in our sample emitted a total of 9.5 million tons of SO_2 during 1990, the year Title IV was passed. By 1995, our 148 plants had reduced their SO_2 emissions by 4.6 million tons from their 1990 levels, cutting them almost in half, although Title IV had only required them to reduce emissions by 15%, to 8.1 million tons.

VI. Distribution of Benefit and Costs

In Table 2 we present two scenarios of health benefits and abatement costs. In Scenario 1 we calculate the benefits and costs associated with the actual 1995 SO_2 emissions reductions (costs are based on Ellerman et al (1997)): counterfactual SO_2 emissions minus actual emissions. The counterfactual emissions in 1995 are those we would have observed in the absence of the CAAA of 1990, based on calculations presented in Ellerman et al (1997). In Scenario 2 we take the actual reduction in SO_2 emissions as given, and compare the costs and benefits associated with achieving that aggregate reduction using two different policy regimes, allowance trading and command-and-control (reducing the allowable emissions rate uniformly across plants), based on calculations from Keohane (2003). A visual comparison of the benefits from reducing SO_2 emissions under the two scenarios can be seen in Figures 2 and 3. Not surprisingly, given the concentration of the plants in the Midwest and the pattern of airflow from west to east, the benefits that result from the large reductions in emissions in Scenario 1 are highly concentrated geographically. Scenario 2 involves a reallocation of emissions reductions across plants, so we see both losers and winners in Figure 3.

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[10] We would like to thank Nat Keohane for providing us with this data.
[11] If we had data on cross-state electricity sales, we could adjust our cost calculations to reflect this.
[12] We would like to thank Denny Ellerman for providing us with this data.

As expected, the aggregate benefits in 1995 resulting from reductions in SO_2 emissions from the 1995 counterfactual levels far outweigh their costs: we estimate benefits of nearly $56 billion and costs of only $558 million. An alternative assumption on abatement costs is that the actual cost of a ton of abatement is equal to the allowance price ($128.5 in 1995), which results in total abatement costs of only $496 million. In either case these increased abatement costs are dwarfed by the increased benefits from the SO_2 reduction, which are roughly 100 times as large.

Scenario 2 shows that allowance trading results in a sizable reduction in abatement costs ($94 million or 16.8%), relative to achieving the same aggregate emissions by a hypothetical command-and-control system. These cost savings are outweighed, however, by the changes on the benefits side. Plants with decreased emissions under allowance trading are more likely to be low-benefit plants, while plants with higher emissions under allowance trading are more likely to be high-benefit plants. In other words, we find that plants which buy allowances (to emit more SO2) are more likely to be high-benefit plants, while plants that sell allowances (and thereby emit less SO2) are more likely to be middle- or low-benefit. This is reflected in the average benefits at buying and selling plants: the buying plants have a mean benefit of $17,519 while the selling plants have a mean benefit of $14,777. These differences are not huge, but it is still the case that the plants which are buying (selling) allowances are those plants which yield the highest (lowest) benefits from abating a ton of SO_2. This result drives the negative impact of the trades on overall benefits observed in Table 2, and suggests that the allowance trading system might benefit from a spatially-based 'exchange rate' based on differences in the impacts of emissions across these plants.

Tables 3A, 3B, and 3C explore in more detail the differences across plants in marginal benefits generated from reductions in SO_2 emissions. Table 3A shows the distribution of the benefits per ton of reduction across our 148 plants. The variation in these numbers across plants is based on a variety of factors, including effective stack height and meteorological conditions, though the principal determinant is the population density downwind. There are a few outliers at the top and bottom of the distribution, but most plants fall between $9,600 and $19,500 per ton in marginal benefits. The plants towards the top of the distribution tend to be in places like Pennsylvania, while plants in Alabama, Florida, Georgia, and Mississippi tend to be near the bottom, although there is some within state variation as well.

Table 3B examines the hypothetical results from Scenario 2 in more detail, comparing plants which had higher emissions under the allowance trading scenario to plants which had higher emissions under the command-and-control scenario. Table 3C contains a similar comparison, but this time we analyze the actual emission decisions of plants, seeing whether the plants are buying or selling allowances in 1995. The two tables give similar results – plants with low marginal benefits tend to be sellers of allowances, while plants with high marginal benefits tend to be buyers of allowances.

What causes these differences across plants in marginal benefits? The largest factor is the location of the plant, but stack height is also important. Table 4 illustrates that there are large differences in marginal benefits across EPA regions. In particular, EPA regions 3 and 5 tend to have more plants with higher marginal benefits, while there are more plants with lower marginal benefits in EPA regions 4 and 7. Table 4 also shows that the very highest marginal benefit plants all have relatively low stacks (under 250 feet in effective stack height). When this is coupled with being located near a metropolitan area, the emissions from the plant can have a relatively strong local effect. Most of the plants in our sample have considerably higher stacks, and such plants tend to have small or moderate marginal benefits. Also note that plants with higher benefits tend to have higher abatement costs. This helps explain the finding that allowance trading has tended to move emissions from low-benefit to high-benefit plants – plants with higher costs are more likely to buy allowances, and the current trading system provides them with no incentive to consider the extent to which their own emissions are likely to be especially harmful. An examination of the data for individual plants shows that large, newer plants with tall stacks with relatively low benefits tend to be doing much of the additional abating required under allowance trading.[13]

We now turn to an examination of the possibilities of spatially-based limits on trading between plants, in order to reduce the number of trades which increase emissions at high-benefit plants and reduce emissions at low-benefit plants. Since marginal benefits are connected to downwind population, which is expected to differ by plant location, one possible solution is to define a set of trading regions and to require that trades occur only between plants in the same region. If plants in the same region have the same marginal benefits, this will rule out problematic trades. Our data does not identify individual trades, but presents aggregate purchases (or sales) for each plant.[14] We can simulate the effect of trading regions by requiring the buying and selling of allowances to balance within each region, and seeing how this affects the aggregate benefits of reducing emissions, assuming that the changes in allowance trading lead to comparable changes in plant-level emissions.

Table 5A shows the distribution of buying and selling within each EPA region, while Table 5B shows the distribution for each state; each table also presents the national totals. As expected, the national-level data show that emissions from the buyers tend to have higher marginal benefits than emissions from the sellers (roughly 10% higher – benefits per ton of \$16,500 vs. \$15,000). We see considerable heterogeneity in the trading behavior and marginal benefits across states within the same region. Most states have some plants buying allowances and some plants selling them, and there is often a considerable difference in marginal benefits between buyers and sellers. We see that some regions have relatively consistent behavior across plants in different states (e.g. region 3 with allowance buying and region 7 with allowance selling in nearly all states of the region), but that others show more heterogeneity across states (e.g. region 4 with allowance selling by plants in Georgia and allowance buying by plants in Kentucky and Tennessee). The key element for the success of a trading zone approach is the distribution of the marginal benefits. The evidence that there is substantial within-region heterogeneity in marginal benefits indicates that trades between high- and low-benefit plants would continue, leading to possible problems for aggregate welfare.

Table 6 shows the results from two simulations of the impact of changing the allowance trading process by imposing trading zones. The first simulation splits the set of plants into groups based on EPA regions. The second creates two 'super-regions', one including regions 4 and 7 (the Southern and Midwestern regions) and the other including the rest of the sample (the Northeast regions).[15] In both cases we force balanced trading within each region. We first

[13] We have also examined the correlations among these variables (available from authors), but this did not add much additional information to the results presented here.

[14] We have recently received the necessary data to identify individual trades – the buying plant, the selling plant, their location, and the total number of allowances traded. This will allow us to do more detailed simulations.

[15] We considered simulating the effects of state-level trading zones, but this ran into the problem that some states have no buyers (or no sellers) of allowances – so there is no natural way to force those states into equilibrium. Creating 22 separate trading zones also raises concerns with implementation in terms of the market power that it would generate for individual facilities within the smaller states.

calculate the excess demand (or supply) for allowances within the region. If there is excess demand, we eliminate it by increasing sales and decreasing purchases of allowances within the region, in proportion to the size of the plants buying and selling allowances within that region (and similarly for excess supply). To the extent that this reduces purchases (or increases sales) by high-benefit plants, it will increase social welfare.

The results show some benefits from trading zones, but they are not very large. The baseline data indicates 867,000 allowances being traded across plants, for which the discrepancy in marginal benefits between buyers and sellers amounts to a shortfall in benefits of $1.055 billion. Imposing the 2-region trading zone model would result in excess demand (supply) of about 25,000 allowances in each region, which reduces the shortfall in benefits by $113 million, or about 11% of the original shortfall. A 6-region trading zone model takes advantage of the greater variation in excess demand and supply across those regions, reducing the shortfall in benefits by $143 million, or about 14% of the original shortfall. While the absolute change in the shortfall from these trading zones might seem large in absolute terms, it would still leave 80-90% of the shortfall in place, and at the cost of considerably complicating the trading process (and possibly losing the political impetus that led to passing the enabling legislation). As noted earlier, the substantial within-region heterogeneity in marginal benefits is limiting the benefits from trading zones.

An alternative approach would be to assign each plant an 'exchange rate' proportional to its marginal benefits, and require that plants buy sufficient allowances to cover their emissions, after accounting for the exchange rate. This would tend to force high-benefit plants to abate their pollution (rather than buying many extra allowances to compensate for the high benefits). Our initial attempts to model an individual plant's actual decision about buying and selling allowances have not been very successful (not predicting very well the actual buy/sell decision), so we are not presenting those results here. We can note that the variation in marginal benefits across plants is somewhat larger than the variation in our measure of abatement costs, so the plants' final decisions about buying and selling allowances under an 'exchange rate' system are likely to be driven primarily by differences in marginal benefits, rather than costs.

VII. Concluding Remarks

In this paper we analyze plant-level information on fossil fuel fired electric utilities to examine the distribution of costs and health benefits associated with the air quality improvement achieved by Title IV of the 1990 CAAA and compare it to the distribution under a command-and-control regime. In addition to comparing the costs and health benefits that arise from reductions in SO_2 emissions under Title IV, we use data on abatement costs to simulate the impact of requiring a comparable reduction in SO_2 emissions under the old command-and-control regime, by assuming uniform emission standards at all plants. We examine the distribution of benefits and costs both in terms of the regions being affected and the socio-economic composition of the affected population.

Our results for Scenario 1 suggest that, as expected, the aggregate health benefits in 1995 caused by reductions in SO_2 emissions under Title IV greatly exceeded their costs. We estimate benefits of $56 billion and costs of only $558 million leading to $55 billion dollars of net benefits from the SO_2 reductions.

Our results for Scenario 2 compare the results from allowance trading under Title IV versus a hypothetical command-and-control system with uniform emission standards that would achieve the same overall reduction. We find that allowance trading saves a substantial fraction of the abatement costs, but the geographic shift in SO_2 emissions induced by allowance trading goes in the other direction, generating a reduction in the abatement benefits. To understand the importance of shifts in emissions across plants for Scenario 2, we examine the distribution of the marginal benefits of reducing emissions across our 148 plants. The differences are not huge: the median benefit per ton is about \$15,000 and 80% of plants fall between \$10,000 and \$20,000. However, when we consider which plants are buying or selling allowances, we find that plants that buy allowances tend to be high-benefit and plants that sell allowances tend to be middle or low-benefit.

This helps explain the negative net benefits from allowance trading we find for Scenario 2, and raises the question of whether a spatially-based approach to trading would improve the results. We find that alternative trading zone models (with 2 and 6 trading zones) result in only modest reductions in the overall performance of the model (reducing the shortfall in benefits by about 11-14%). This arises from the considerable heterogeneity of marginal benefits across plants within the same region. Given the necessary increase in complexity for the trading system, the modest improvements may not be sufficient justification for making a change. Next steps in the evolution of this research will involve incorporating more detailed measures of abatement costs and data on actual individual allowance trades to generate a plant-level (or unit-level) model of the tradeoff between abatement costs and allowance purchases, allowing us to model the impact of marginal benefit-based exchange rates on the overall performance of the allowance trading system.

REFERENCES

Abt Associates, "The Particulate-Related Health Benefits of Reducing Power Plant Emissions," prepared for Clean Air Task Force (October 2000).

Atkinson, S.E. and T.H. Tietenburg, "The Empirical Properties of Two Classes of Designs for Transferable Discharge Permit Markets," *Journal of Environmental Economics and Management*, **9**, 101-121 (1982).

Baumol, W. J., and W.E. Oates, *The Theory of Environmental Policy*, Cambridge University Press, 1988.

Burtraw, D., A. Krupnick, E. Mansur, D. Austin, and D. Farrell, "The Costs and Benefits of Reducing Acid Rain," Resources for the Future Discussion Paper 97-31-REV, Washington, D.C.: Resources for the Future (1997).

C.P. Carlson, D. Burtraw, M. Cropper, and K. Palmer, "SO2 Control by Electric Utilities: What are the Gains from Trade?," *Journal of Political Economy* **108**, 1292-1326 (2000).

K. Chay and M. Greenstone, "The Impact of Air Pollution on Infant Mortality: Evidence from Geographic Variation in Pollution Shocks Induced by a Recession," Quarterly Journal of Economics, 1121-1167 (2003a).

____________, "Air Quality, Infant Mortality, and the Clean Air Act of 1970," NBER Working Paper 10053 (2003b).

A. D. Ellerman, R. Schmalensee, P. Joskow, J. Montero, and E. Bailey, Emissions Trading Under the U.S. Acid Rain Program: Evaluation of Compliance Costs and Allowance Market Performance., MIT Center for Energy and Environmental Policy Research, 1997.

A. D. Ellerman, and J. Montero, "The Declining Trend in Sulfur Dioxide Emissions: Implications for Allowance Prices," *Journal of Environmental Economics and Management*, **36**, 26-45 (1998).

A.D. Ellerman, "Lessons from Phase 2 Compliance with the U.S. Acid Rain Program," presented at the EPA's Market Mechanisms and Incentives: Applications to Environmental Policy (May, 2003).

F. M. Gollop, and M.J. Roberts, "Cost Minimizing Regulation of Sulfur Emissions: Regional Gains in Electric Power," *Review of Economics and Statistics*, 81-90 (1985).

P.L. Joskow, R. Schmalensee, and E. Bailey, "The Market for Sulfur Dioxide Emissions," *The American Economic Review* **88**, 669-685 (1998).

N. Keohane, "Environmental Policy and the Choice of Abatement Technique: Evidence from Coal-Fired Power Plants." Working paper available at www.som.yale.edu/faculty/nok4/files/papers/so2.pdf. (2002)

N. Keohane, “What Did the Market Buy? Cost Savings Under the U.S. Tradable Allowances Market Program for Sulfur Dioxide”, Yale Center for Environmental Law and Policy Working Paper ES #33 (2003).

D. A. Latimer, “Particulate Matter Source-Receptor Relationships Between All Point and Area Sources in the United States and PSD Class I Area Receptors,” prepared for EPA, OAQPS (September 1996).

Pope III, C. A., Burnett, R. T., Thun, M. J., Calle, E. E., Krewski, D., Ito, K., Thurston, G. D. (2002). Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. *Journal of the American Medical Association*, **287**, 1132–1141.

D. Popp, “Pollution Control Innovations and the Clean Air Act of 1990,” *Journal of Policy Analysis and Management* **22**, 641-660 (2003).

R. Schmalensee, P.L. Joskow, A.D. Ellerman, J.P. Montero, E.M. Bailey, “An Interim Evaluation of Sulfur Dioxide Emissions Trading,” *Journal of Economic Perspectives* **12**, 53-68 (1998).

R. Shadbegian, W. Gray, and J. Levy, “Spatial Efficiency of Pollution Abatement Expenditures,” presented at the Western Economic Association Meetings (June 1999), NBER Environmental Economic Meetings (April 2000), International Atlantic Economic Association Meetings (October 2000), and U.S. E.P.A. National Center for Environmental Economics (October 2000).

R. Shadbegian and C. Morgan, “Environmental Regulation in the Electric Utility Industry: Are There Productivity Gains From Emissions Trading?” Presented at the International Atlantic Economic Association Meetings (October 2002) and the Western Economic Association Meetings (July 2003).

R. Shadbegian, W. Gray, and C. Morgan, “Benefits and Costs From Sulfur Dioxide Trading: A Distributional Analysis” in Acid in the Environment: Lessons Learned and Future Prospects (ed.s Gerald R. Visgilio and Diana M. Whitelaw), Springer Science+Media (forthcoming 2006).

Tietenberg, T., “Tradeable Allowances for Pollution Control When Emission Location Matters: What Have We Learned?” *Environmental and Resource Economics*, March 1995.

U.S. Environmental Protection Agency, "The Benefits and Costs of the Clean Air Act, 1970 to 1990," October, 1997.

U. S. Environmental Protection Agency, EPA Acid Rain Program 2001 Progress Report, http://www.epa.gov/airmarkets/cmprpt/arp01/2001report.pdf, November 2002.

Table 1 – Phase I Plants

	Phase I Plants*
SO_2 Emissions in 1990 (tons)	9,468,183
SO_2 Emissions in 1995 (tons)	4,902,778
Allowances in 1995	8,076,472
Boilers	374
Plants	148

* = Includes the 110 Table A plants plus the 38 “Substitution and Compensation” plants

Table 2 – Benefits and Costs

	Scenario 1	**Scenario 2**
Benefits	$55.94 billion	-$1,255 million
Costs	$0.56 billion	-$94 million
Net Benefits	$55.38 billion	-$1,161 million

Table 3A – Distribution of Benefits per Ton Reduction Across Plants

Distribution	Benefits/Ton
Maximum	$35,868
90%	$19,662
75%	$17,477
50%	$15,414
25%	$12,575
10%	$9,601
Minimum	$3,763

Table 3B – Distribution of Benefits per Ton Reduction (Scenario 2 Outcomes)
Command-and-Control vs. Allowance Trading

	Low Benefits (<$12,500)	Middle Benefits ($12,500-$17,500)	High Benefits (>$17,500)
Higher Emissions under Allowance Trading	**9**	**34**	**20**
Lower Emissions under Allowance Trading	**20**	**32**	**5**

Table 3C – Distribution of Benefits per Ton Reduction
Actual Trading Outcomes - Buying and Selling

	Low Benefits (<$12,500)	Middle Benefits ($12,500-$17,500)	High Benefits (>$17,500)
Allowance Buyers	**12**	**36**	**15**
Allowance Sellers	**19**	**28**	**9**

Table 4 – Determinants of Benefits per Ton Reduction

	Low Benefits (<$12,500)	Middle Benefits ($12,500-$17,500)	High Benefits (>$17,500)
Region			
1 (MA,NH)	**1**	**0**	**1**
2 (NJ,NY)	**2**	**3**	**1**
3 (MD,PA,WV)	**0**	**13**	**10**
4 (AL,FL,GA,KY,MS,TN)	**22**	**11**	**1**
5 (IL,IN,MI,MN,OH,WI)	**4**	**43**	**13**
7 (IA,KS,MO)	**12**	**7**	**1**
Stack Height			
Low	**2**	**12**	**14**
Medium	**17**	**24**	**13**
High	**22**	**41**	**3**
Abatement Costs			
Low	**20**	**33**	**7**
Medium	**15**	**22**	**10**
High	**6**	**22**	**13**

Table 5A – Distribution of Buying and Selling Across EPA Regions

Region	Total	Buy	Sell	Total Buy	Total Sell	Net Buy	MB-Buy	MB-Sell
1	2	1	1	4612	-1848	2764	$18,155	$9,510
2	6	2	2	7791	-48537	-40746	$17,593	$10,366
3	23	14	7	199284	-156723	42561	$18,229	$20,962
4	34	16	10	277268	-225112	52156	$12,545	$11,332
5	63	27	26	371025	-350174	20851	$17,584	$17,330
7	20	3	10	6915	-84499	-77584	$18,441	$9,814
	148	63	56	866893	-866893	0	$16,498	$14,982

Table 5B – Distribution of Buying and Selling Across States

Region	State	Total	Buy	Sell	Total Buy	Total Sell	Net Buy	MB-Buy	MB-Sell
1	MA	1	0	1	0	-1848	-1848	-	$9,510
1	NH	1	1	0	4612	0	4612	$18,155	-
2	NJ	1	1	0	1161	0	1161	$19,507	-
2	NY	5	1	2	6629	-48537	-41908	$15,679	$10,366
3	MD	4	3	1	21347	-1837	19510	$18,517	$28,203
3	PA	12	7	3	86575	-27997	58578	$18,978	$19,057
3	WV	7	4	3	91362	-126889	-35527	$16,703	$20,453
4	AL	3	1	1	6743	-19045	-12302	$11,826	$9,324
4	FL	3	2	0	11668	0	11668	$8,283	-
4	GA	10	2	5	1728	-124781	-123053	$10,198	$10,928
4	KY	12	7	2	141832	-11484	130348	$15,196	$15,518
4	MS	2	1	1	9515	-431	9084	$5,588	$5,749
4	TN	4	3	1	105783	-69371	36412	$13,324	$12,575
5	IL	12	5	5	87372	-48005	39367	$14,848	$15,998
5	IN	15	11	4	147839	-26129	121710	$15,754	$18,249
5	MI	2	1	1	812	-16234	-15422	$30,354	$16,393
5	MN	2	0	1	0	-15	-15	-	$15,371
5	OH	22	9	8	134523	-180352	-45829	$20,195	$19,436
5	WI	10	1	7	478	-79439	-78961	$15,128	$15,762
7	IA	6	1	3	1543	-1725	-182	$4,322	$12,061
7	KS	2	0	1	0	-3636	-3636	-	$3,931
7	MO	12	2	6	5372	-79138	-73766	$25,500	$9,671
	TO	148	63	56	866893	-866893	0	$16,498	$14,982

Table 6 – Shortfalls in Benefits from Allowance Trading
Impacts of Trading Zones

Excess demand/supply	Shortfall in Benefits	$ Improvement over Baseline	% Improvement over Baseline
Baseline model (no zones)			
0	-$1055 M	$0	0%
2-region model (region 4+7, 1+2+3+5)			
(25429, -25429)	-$942 M	$113 M	10.7%
6-region model (regions 1,2,3,4,5,7)			
(2764, -40746, 42561, 52156, 20851, -77584)	-$912 M	$143 M	13.6%

Figure 1
Distribution of Plants in Database
(148 Plants; scale=1995 SO_2 emissions in tons)

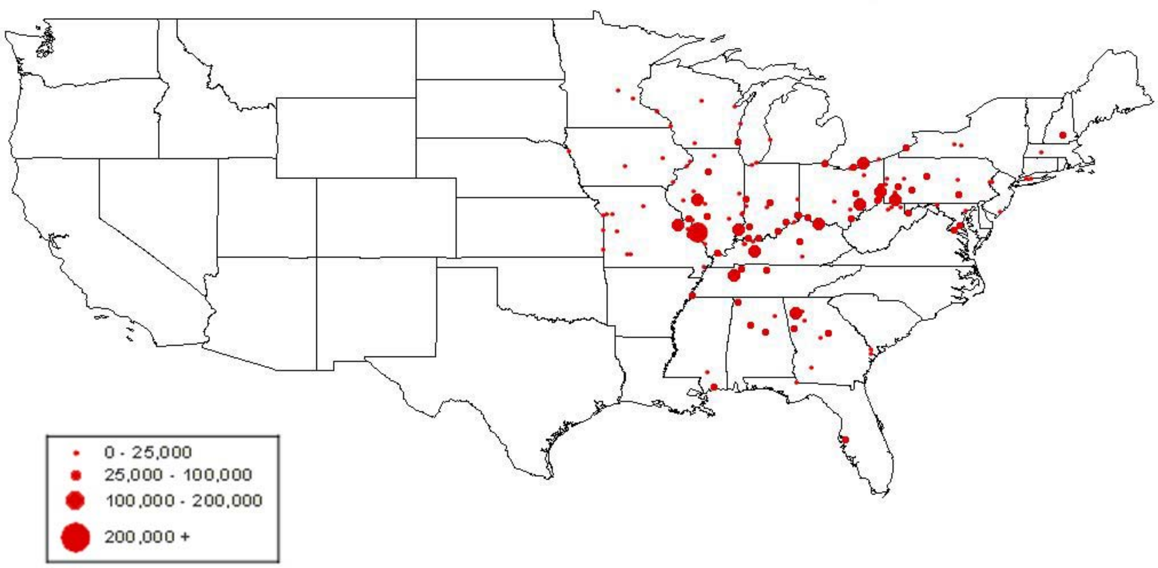


Figure 2
Geographic Distribution of Benefits
Scenario 1

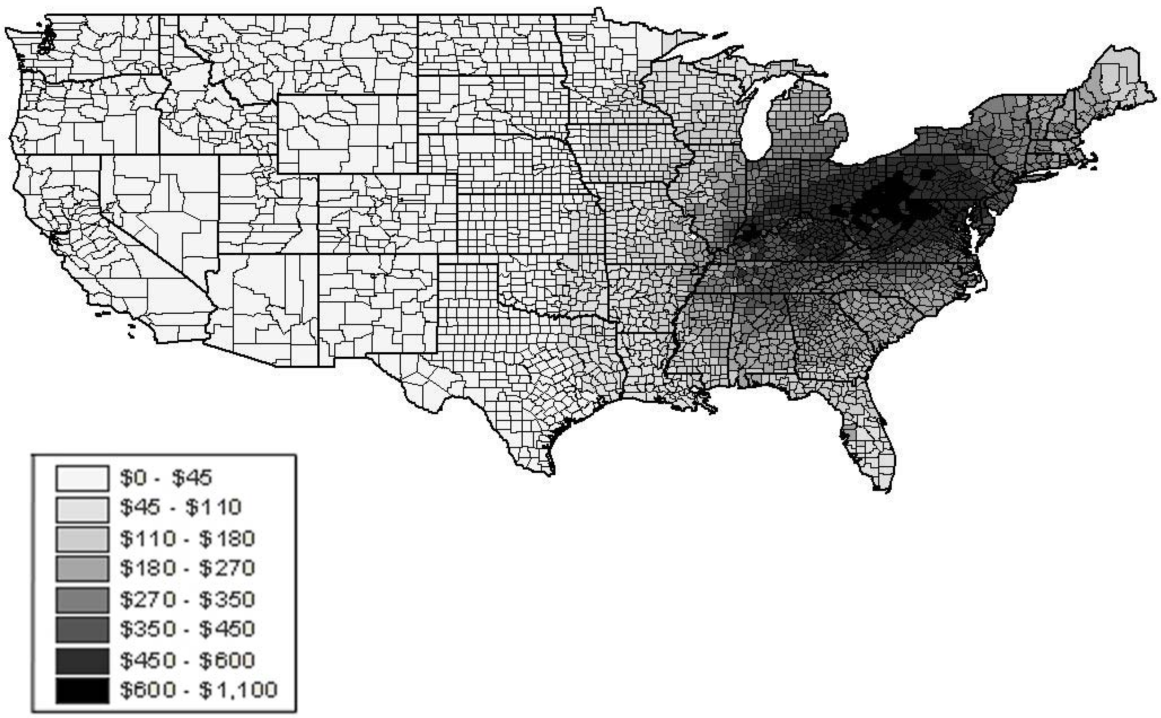
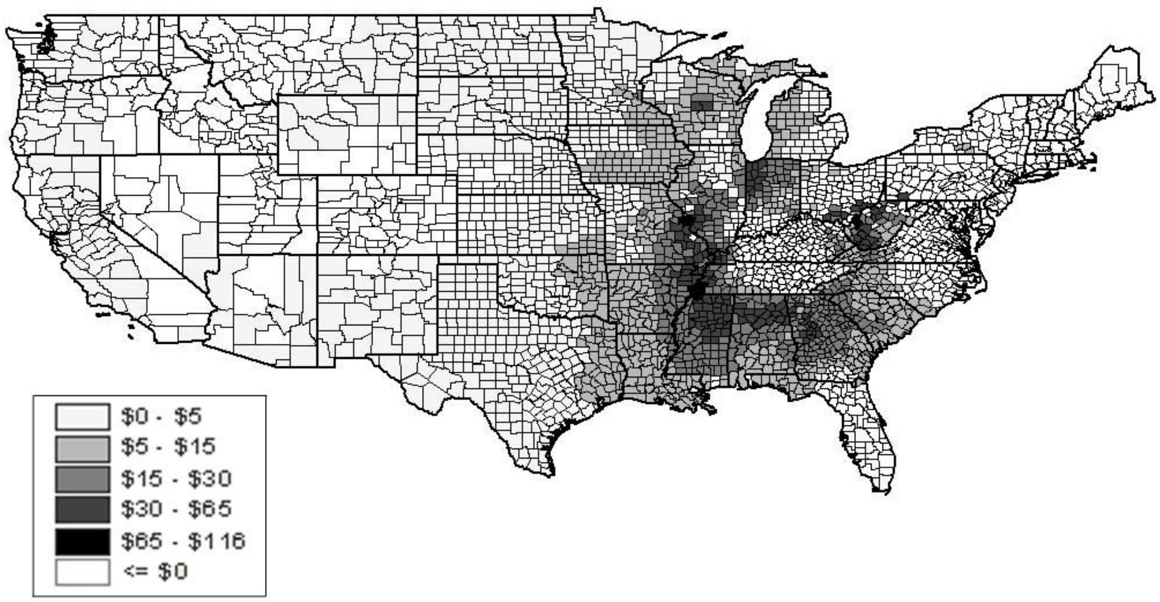
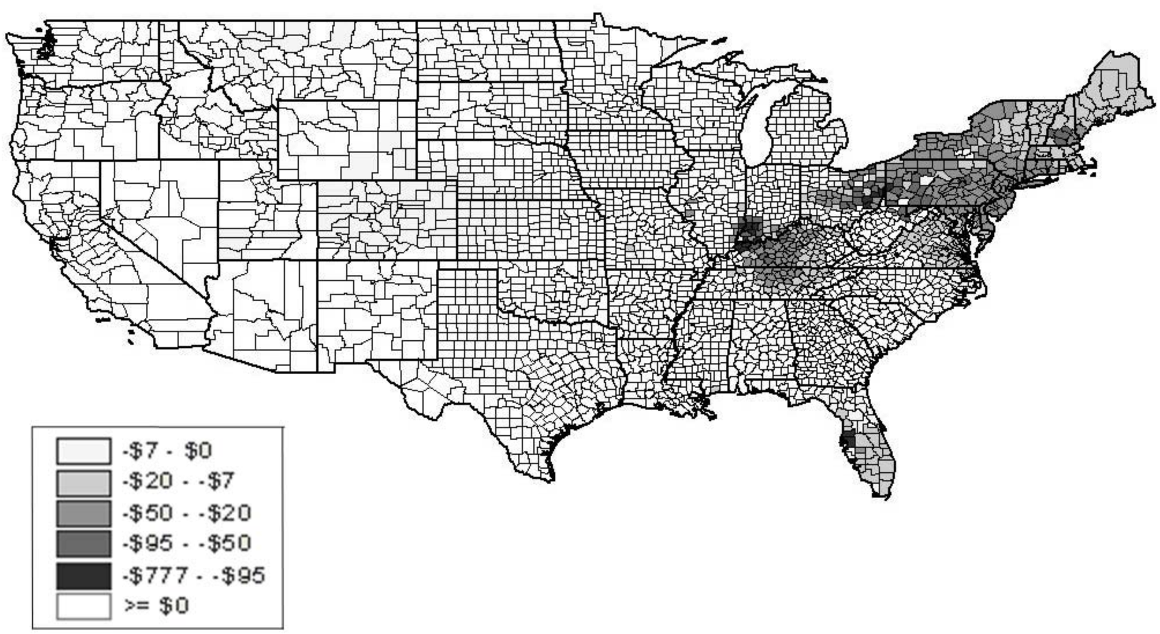


Figure 3
Geographic Distribution of Benefits
Scenario 2

Net Winners



Net Losers



Emissions Trading, Electricity Industry Restructuring, and Investment in Pollution Abatement*

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Abstract

Policy makers are increasingly relying on emissions trading programs to address environmental problems caused by air pollution. If polluting firms in an emissions trading program face different economic regulations and investment incentives in their respective industries, emissions markets may fail to minimize the total cost of achieving pollution reductions. This paper analyzes an emissions trading program that was introduced to reduce smog-causing pollution from large stationary sources (primarily electricity generators). Using variation in state-level electricity industry restructuring activity, I identify the effect of economic regulation on pollution permit market outcomes. There are two important findings. First, plants in states that have restructured electricity markets were less likely to adopt more capital intensive compliance options. Second, this economic regulation effect, together with a failure of the permit market to account for spatial variation in marginal damages from pollution, have had substantial negative health impacts.

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When the U.S. federal government first began regulating major sources of air pollution in the 1960s, the conventional approach to meeting air quality standards involved establishing maximum emissions rates or technology-based standards for regulated stationary sources. At that point, the idea of establishing a cap on total permitted emissions, distributing tradeable pollution permits to regulated sources, and letting a market coordinate pollution reduction among regulated firms was just beginning to take hold among a small group of economists (Coase, 1960; Crocker, 1966; Dales, 1968; Baumol and Oates, 1971). Over the past few decades, the environmental regulatory landscape has changed dramatically. The "cap and trade" approach to regulating point sources of pollution is now the centerpiece of industrial air pollution regulation in the United States.

Economists have long pointed out that an efficient pollution permit market minimizes the total social cost of meeting an exogenously determined cap on emissions. In the first-best permit market equilibrium, each firm chooses a level of pollution abatement such that the marginal cost of reducing pollution is set equal to the social marginal benefit from emissions reduction at the firm. However, there are two important assumptions underlying economic arguments for the efficiency of permit markets that are unlikely to be satisfied by many existing and proposed cap and trade (CAT) programs.[1] The first pertains to the objectives of the firms regulated under CAT programs; the second to the terms of permit trading. This paper assesses the consequences of violating these two assumptions in practice using a unique data set from a major U.S. Nitrogen Oxide (NOx) emissions trading program (the NOx Budget Trading Program). I find that inter-state variation in economic regulation, together with the failure of the permit market to account for spatial variation in marginal damages from pollution, have distorted investment in pollution controls away from the first-best, thereby reducing the efficiency of pollution permit market outcomes.

In a formal proof of the existence of a cost effective permit market equilibrium, it is typical to assume that all firms have the same objective function (Montgomery, 1972). Although firms are assumed to differ in terms of the price they receive for their products, costs of production, and costs of reducing emissions (indeed, it is this heterogeneity that gives rise to gains from permit

[1] Several assumptions are required to demonstrate the efficiency of cap and trade programs. These include: zero transaction costs, perfectly competitive permit markets, perfect enforcement and compliance, perfectly competitive product markets and profit maximizing (or cost minimizing) behavior. In a multiple-receptor, non-uniformly mixed pollutant case, economists further assume an "exposure" or damage based permit system.

trading), it is assumed that all firms are essentially solving the same cost minimization problem when deciding how to comply with CAT regulation.

In fact, firms in the same pollution permit market may approach the choice of how to comply with a CAT program very differently. The vast majority of the emissions regulated under CAT programs come from electricity generators.[2] The recent wave of electricity industry restructuring in the United States has resulted in significant inter-state variation in electricity industry economic regulation. Thus, in addition to having different production and abatement costs, generators in the same CAT program face different economic regulation and investment incentives depending on the nature of their electricity market.

The first question addressed by this paper: have differences in electricity market regulation affected how coal plant managers chose to comply with a multi-state NOx emissions trading program?[3] I develop and estimate a random-coefficients logit (RCL) model of the firm's compliance choice that controls for unit-level variation in compliance costs and allows for correlation across choices made by the same decision maker. I find that plants in restructured electricity markets were less likely to choose more capital intensive compliance options as compared to similar plants operating in regulated electricity markets. More capital intensive compliance options are associated with significantly greater emissions reductions.

These findings have implications for both technical and allocative efficiency. With respect to the former, these results imply that it is not always the plants with the lowest NOx control costs that have invested in pollution control equipment. Observed compliance decisions are compared to those predicted by a deterministic model which minimizes the total technology hardware and operating costs required to comply with the cap. Results suggest that too much investment has occurred in regulated versus restructured electricity markets, as compared to the relative levels of investment predicted by the deterministic model. Unfortunately, because of relatively poor air quality in states with restructured electricity markets, these are precisely the states where

[2] All of the emissions regulated under the Acid Rain Program and over 90% of the emissions regulated under the NOx SIP Call come from electricity generators. The cap and trade program laid out in the proposed Mercury Rule applies exclusively to the electricity sector.

[3] The paper focuses exclusively on the compliance decisions of coal-fired electricity generators. 85 percent of the point source NOx emissions regulated under the program comes from coal plants.

pollution control equipment could deliver the greatest health benefits.

These results are particularly troubling because pollution permit markets, as they are currently designed, fail to reflect considerable spatial variation in marginal benefits from pollution reductions. Currently, all major cap and trade programs are "emissions-based": a permit can be used to offset a unit of pollution, regardless of where in the program region the unit is emitted. Designing a program in this way presumes that the health and environmental damages resulting from the permitted emissions are independent of where in the regulated region the emissions occur. A growing body of scientific evidence indicates that this is not the case for NOx, which is classified as a "non-uniformly mixed" pollutant because damages from increased NOx emissions depend on the location of the source (Lin et al., 2002; Mauzerall et al., 2005).

This leads to the second key assumption underlying the efficiency of permit market equilibria that is often violated in practice. Economists have traditionally assumed that CAT programs regulating non-uniformly mixed pollutants will be "exposure-based" (i.e., permits will be defined in terms of units of damages) rather than emissions-based (Montgomery, 1972; Tietenberg, 1974). In the second part of the paper, I evaluate the consequences of violating this assumption in a case where inter-state variation in electricity market regulation has the potential to exacerbate the allocative inefficiency associated with emissions-based trading. The estimates of the RCL compliance choice model are used to assess whether an exposure-based market design would have significantly affected the spatial distribution of NOx emissions permitted under the NOx Budget Trading Program (NBP). I derive parameters of conditional distributions specific to each decision maker (i.e. plant manager or parent company). Drawing from these conditional distributions, I predict the compliance choices that these agents most likely would have made had the NOx emissions market been designed to reflect spatial heterogeneity in marginal damages from pollution.

I find that the decision to adopt an emissions-based program (versus a damage-based permit market designed to achieve the same total emissions) has substantially increased daily NOx emissions in areas where air quality problems are most severe. Epidemiological studies consistently find a statistically significant association between NOx related air quality problems and increased mortality and morbidity (WHO, 2003). Simulation results suggest that exposure-based permit trading would have moved as much as 300 tons of NOx per day out of high damage areas and into low damage areas where the pollution does less damage.[4] Recent epidemiological research suggests that a spatial shift in NOx emissions of this magnitude could reduce premature deaths from ozone exposure by hundreds each year (Mauzerall et al. 2005).

These findings are relevant to three related areas of the literature. First, a number of authors have addressed the broad question: how effective are existing U.S. cap and trade programs? Most have focused exclusively on the Acid Rain Program (ARP) that was established in 1990.[5] This is, to my knowledge, the first paper to evaluate the performance of the NBP, which is second only to the ARP in terms of size and scope.

Second, strands of both the industrial organization and environmental economics literatures have considered the effects of economic regulation and industry structure on firms' investment decisions.[6] Previous empirical work that considers how economic regulation in electricity markets has affected firms' CAT compliance choices has focused predominantly on the Acid Rain Program.[7] Because the Acid Rain Program started before restructuring began, these papers use more subtle variations in cost recovery rules and coal protection measures to identify an effect of electricity market regulation on compliance choices. Results have been mixed.[8] I revisit this question post-restructuring, now that there is significantly more interstate variation in electricity industry regulation and investment incentives, and thus increased potential for variation in economic regulation to undermine the efficiency of the permit market.

Finally, there is a growing literature that considers non-uniformly mixed pollution permit

[4] This daily shift in NOx emissions would only occur during "ozone season" (May-September) when the the NOx SIP Call is in effect. Firms do not need to purchase permits to offset uncontrolled emissions occuring outside ozone season because NOx related air quality problems are less severe during the cooler months of the year.

[5] Papers analyzing the operation and performance of the Acid Rain Program include: Joskow et al.(1998), Keohane (2005), Shadbegian et al. (2006), Schmalansee et al.(1998), and Stavins (1998).

[6] There is a large literature that extends, corrects and tests the "Averch and Johnson effect" (1962). Empirical results have been mixed. In the environmental economics literature, several papers have illustrated how, in theory, economic regulation can undermine the ability of a pollution permit market to operate efficiently (see Bohi and Burtraw, 1992; Carlson et al., 1998; Coggins and Smith, 1993; Fullerton et al., 1997).

[7] Mansur (2004) is an exception. He considers how market concentration in restructured electricity markets affects firms' short run compliance decisions under market-based NOx regulation.

[8] Bailey (1998) tests whether permit market participation (measured at the state level) is affected by how favorable an electricity market regulator has been to shareholder interests. She finds very limited evidence. Keohane (2005) finds no discernable effect of economic regulation on the decision to install a scrubber. Conversely, Arimura (2002) and Sotkiewicz (2003) do find evidence that economic regulations affected ARP compliance decisions.

trading.[9] Previously, deterministic models of the compliance decision that assume strict cost minimization on behalf of all firms have been used to assess ex ante the merits of imposing spatial constraints on NOx permit trading.[10] The analysis presented here allows for a more realistic ex post evaluation of alternative, exposure-based permit market designs. Unlike previous studies, I find that the adoption of exposure-based NOx permit trading would have delivered significant health benefits. This result is particularly relevant to the debate that is currently taking place over the design of future emissions trading programs.[11]

The next two sections describe the emissions trading program, electricity market regulation, and restructuring in the United States. Section 3 describes the data and presents summary statistics. Section 4 introduces a model of the firm's compliance decision. Estimation results are presented in Section 5. In Section 6, I use the model to simulate compliance decisions under exposure-based trading. Section 7 concludes.

1. The NOx Budget Program

The NOx Budget Program (NBP) is an emissions trading program that limits emissions of NOx from large stationary sources in nineteen Eastern states. These NOx emissions contribute to the formation of ozone.[12] High ambient ozone concentrations have been linked to increased mortality, increased hospitalization for respiratory ailments, irreversible reductions in lung capacity, and ecological damages.

The NBP was primarily designed to help Northeastern states come into attainment with

[9] Analytical papers that consider imposing spatial constraints on trading and related alternative market designs include Duggan and Roberts (2002), Hahn (1990), and Krupnick et al. (1983). Shadbegian et al.(2006) use data from the first year of the ARP to assess the benefits from limiting permit trading to within pre-determined zones. They conclude that considerable heterogeneity in marginal benefits within regions would limit the potential gains from such a system.

[10] Farrell et al. (1999) consider imposing geographic constraints on NOx permit trading in the Northeast and conclude that the benefits do not justify the costs. Krupnick et al.(2000) argue that there is no clear benefit to spatially differentiated NOx trading. Finally, the EPA considered imposing restrictions on interregional trading during the planning stages of the NBP. The Integrated Planning Model (IPM), a deterministic model that does not reflect interstate variation in electricity market regulation, and which assumes plant managers select compliance options to minimize costs, was used to simulate outcomes under different policy designs. Results suggested that benefits from exposure based trading would be negligible (EPA, 1998c).

[11] In March of 2005, the EPA issued two new, large scale emissions trading programs, both of which regulate non-uniformly mixed pollutants and are emissions-based. One of these programs, the Mercury Rule, has been particularly controversial because the proposed market fails to reflect spatial variation in damages from pollution.

[12] NOx reacts with carbon monoxide and volatile organic compounds (such as hydrocarbons and methane) in the presence of sunlight to form ozone in the lower atmosphere.

the Federal ozone standards. During high ozone episodes, significant portions of the Northeast and parts of the Midwest can fail to attain the Federal standard (OTAG, 1997). Surface ozone concentrations are a function of both in situ ozone production and pollutant transport; both are significantly affected by prevailing meteorological conditions. Several states that were in attainment with Federal ozone standards were included in the NBP because their NOx emissions contribute to the non-attainment problems of downwind states. Although some states contribute significantly more than others to the ozone non-attainment problem, the NBP applies uniform stringency across all 19 states.

The NBP mandated a dramatic reduction in average NOx emissions rates.[13] In the period between when the rule was upheld by the US Court of Appeals (March 2000) and the deadline for full compliance (May 2004), firms had to make costly decisions about how to comply with this new regulation.[14] To comply, firms can do one or more of the following: purchase permits to offset emissions exceeding their allocation from other firms, install one of several types of NOx control technology, or reduce production at dirtier plants during ozone season.[15]

Two factors that are likely to significantly influence a manager's choice of compliance strategy are the up-front capital costs K and anticipated variable compliance costs v (i.e. compliance costs incurred per unit of electricity produced). The capital costs, variable operating costs, and emissions reduction efficiencies associated with different compliance alternatives vary significantly, both across NOx control technologies and across generating units with different technical characteristics.

Figure 1 is a graphical illustration of the compliance choice set corresponding to one particular unit in the sample. Each of the eight points plotted in fixed cost ($/kW) variable cost (cents/kWh) space corresponds to a different compliance "strategy". With the exception of the "no retrofit"

[13] Pre-retrofit emissions rates at affected coal plants were, on average, three and a half times higher than the emissions rate on which the aggregate cap was based (0.15 lbs NOx/mmbtu).

[14] Coal plants in 9 Northeastern states had to achieve compliance by May 2003. Plants in the Southeastern states had to comply by May 31 2004.

[15] The specific control technologies available to a given unit vary across coal-fired units of different vintages and boiler types. Compliance options that incorporate Selective Catalytic Reduction (SCR) technology can reduce emissions by up to ninety percent. NOx emissions rates can be reduced by thirty-five percent through the adoption of Selective Non-Catalytic Reduction Technology (SNCR). Pre-combustion control technologies such as low NOx burners (LNB) or combustion modifications (CM) can reduce emissions by fifteen to fifty percent, depending on a boiler's technical specifications and operating characteristics.

option (i.e. the firm will rely entirely on the permit market to comply with the program), all of the compliance strategies involve retrofitting the unit with a NOx control technology or combination of technologies.[16] Variable costs v include the costs of operating the control technology plus the costs of purchasing permits to offset uncontrolled emissions.[17] The broken line represents a quadratic frontier or envelope function $K(v)$ fit to the points in this choice set that minimize K given v. Points to the right of the fronteir are not cost minimizing.

Choice sets, variable costs, capital costs and emissions reductions associated with a given strategy vary significantly across units with different operating characteristics. For all units, however, the most capital intensive compliance options (i.e., those incorporating selective catalytic reduction technology) are associated with significantly greater emissions reductions.

2. Electricity Industry Restructuring and the Compliance Decision

Until the mid-1990s, over ninety percent of electricity in the United States was generated by vertically integrated investor-owned utilities (IOUs), most of whom were operating as local monopolies regulated by state public utility commissions (PUCs) (Fabrizio et al., 2006). The remainder was supplied by government entities or cooperatives. Traditionally, the most widely used form of regulation has been "rate of return" regulation; rates are set by regulators so as to allow the utility to recover prudently incurred operating costs and earn a "fair" rate of return on its rate base (i.e. the value of assets less depreciation).

Averch and Johnson (1962) illustrate how, under certain conditions, a firm subject to rate of return regulation will find it profitable to employ more capital relative to variable inputs (including labor and fuel) than is consistent with cost minimization. A significant share of the regulation literature has since been devoted to elaborating upon and testing this result.[18] Partly in response

[16]In generating this figure, I implicitly assume that this unit will comply perfectly with the program and that the unit will not achieve compliance by reducing production. Because all units are equipped with continuous emissions monitoring equipment, it is reasonable to assume full compliance. Compliance among coal-fired units was 100 percent in 2004 (EPA, 2005). The assumption that production levels at these coal-fired units will not be significantly affected by this environmental regulation also finds empirical support. This assumption is discussed in detail in Section 6.3.

[17]Using detailed unit-level data, estimates of capital costs and variable compliance costs can are generated for each unit, for each NOx control technology. These calculations assume a permit cost of $2.25/lb NOx; the average futures permit price (per lb NOx) in the years leading up to the NBP. Permits started trading in early 2001 in anticipation of the NBP. A discussion of how these cost estimates are generated is included in Section 4.

[18]Joskow(1974) provides an excellent survey of the earlier Averch and Johnson(AJ) literature. Attempts to

to the debate over the AJ capital bias, "incentive" or "performance based" regulation became increasingly common throughout the 1970s, 1980s and early 1990s.[19]

Proponents of electricity industry restructuring have argued that replacing rate hearings and fuel adjustment clauses with the discipline of a competitive market would increase efficiency and reduce electricity prices. In the 1990s, all fifty states held hearings to assess the benefits of restructuring. Ownership structure and operating incentives have dramatically changed in the nineteen states that have passed restructuring legislation. Utilities in these states have been required or encouraged to divest the majority of their thermal generation assets to unregulated entities. Generators submit bids (prices and quantities) that they are willing to produce in a given hour; Independent System Operators (ISOs) combine these bids and intersect the aggregate supply curve with demand in order to determine the wholesale market clearing price.

2.1. Environmental Compliance Choices in Regulated Electricity Markets

In regulated electricity markets, the environmental compliance decisions of regulated firms were likely influenced by PUC regulations governing capital and variable cost recovery. In each of the seven states in the NBP that have not enacted electricity industry restructuring, firms have successfully sought rate base adjustments in order to recover costs of capital required to invest in NOx control equipment, and to allow shareholders to earn a return on equity.[20] Firms have also won approval for various kinds of rate adjustment clauses or rate freezes which allow them to recover costs associated with purchasing NOx permits, operating pollution control equipment, and pre-approved construction work in progress.[21]

2.2. Environmental Compliance Choices in Restructured Markets

empirically test the AJ effect using data from the US electricity industry have met with mixed results. Courville (1974), Spann (1974) and Hayashi and Trapani (1976) find support for the hypothesis, whereas Boyes (1976) does not.

[19]"Performance based regulation" is a broadly defined concept that refers to any regulatory mechanism that links profits to desired performance objectives (such as improved operating efficiency, improved environmental performance or cost minimizing procurement). Ratemaking under PBR is typically a two-step process. First, a rate base is established to allow the utility to earn a fair rate of return on prudently incurred and projected costs. Second, the utility is given financial incentives to reduce operating costs and increase production efficiency.

[20]In a recent survey, regulators report allowing up to three additional points on the return of shareholder equity for investment in pollution reduction equipment at coal plants, in addition to what would otherwise be earned on prudent investments (NARUC 2004).

[21]For details on PUC rulings in these case, see: Charleston Gazette, 2004; Electricity Daily, 2003; Megawatt Daily, 2003; NARUC, 2004; Platts Utility and Environment Report 1999, 2000a, 2000b, 2001a, 2001b, 2002a, 2002c, 2002d, 2002f; PR Newswire, 2002; Southeast Power Report, 2000.

In the absence of a regulator willing to guarantee cost recovery, the consequences of making large capital investments in pollution control equipment were highly uncertain in restructured electricity markets. The introduction of the NBP increased wholesale prices in restructured electricity markets through its effect on the variable (per kWh) compliance costs of the price-setting or "marginal" generating units. Because coal-fired units typically have low operating costs relative to other units supplying the market, they are typically inframarginal.[22] The generating units that most often set the wholesale electricity price (gas and oil plants) tend to have significantly lower environmental compliance costs as compared to coal. Managers of coal units in restructured electricity markets likely anticipated that the NBP-induced increases in average wholesale electricity prices would not fully reflect their relatively high environmental compliance costs. As one industry analyst has observed "coal plants will still be dispatched, but their (profit) margins will be less."[23]

When there is uncertainty about electricity market conditions, compliance strategies that rely to a significant extent on purchasing permits (versus making large, irreversible capital investments) have option value. If a manager chooses to rely on the permit market for compliance, she has more control over the environmental compliance costs she will incur going forward.[24] This option value did not exist in regulated electricity markets in which firms are guaranteed to recover compliance costs.

Finally, higher costs of capital made securing financing for a large capital investment in NOx control technology relatively more costly for firms in restructured electricity markets (Business Wire 2003; Platts Utility Environment Report, 2002e). Credit rating changes in the energy sector were overwhelmingly negative over the time period in which plant managers were having to make their compliance decision.[25] This negative trend has affected generators operating in restructured industries disproportionately.

[22] A unit will generally operate when its marginal costs of production are less than or equal to the last unit dispatched to serve the load. Because coal-fired units typically have low operating costs relative to other units, they are normally operated to serve the minimum load of a system. They run continuously and produce electricity at an essentially constant rate. Increases in variable environmental compliance costs at these "base load" plants will not significantly affect the wholesale electricity price or the plants' capacity factors.

[23] "High Coal Costs Put the Squeeze On Power Plants."Matthew Dalton; *The Wall Street Journal*; June 29, 2005.

[24] For example, in hours when electricity prices are too low to allow variable compliance costs to be recovered, the firm can choose not to operate.

[25] Downgrades outnumbered upgrades 65 to 20 in 2000; that ratio was up to 182 to 15 in 2002. In 2003, 18 percent of firms were non-investment grade (Senate Committee on Energy and Natural Resources, 2003).

2.3 Generating A Testable Hypothesis

The hypothesis that the type of electricity market in which a coal plant is operating will significantly affect the choice of how to comply with the NBP follows directly from the preceding discussion of industry regulation and investment incentives. A more formal economic model of the relationship between economic regulation and environmental compliance is included in Appendix A. The assumptions underlying the model (namely that plant managers choose compliance strategies to minimize costs) may be too restrictive for this particular application.[26] The model is presented as a possible but not necessary motivation for the empirical analysis that follows.

2.4. Identifying an Effect of Economic Regulation on the Compliance Decision

Ideally, in the interest of empirically testing for a relationship between economic regulation and the environmental compliance decision, coal units would be randomly assigned to either a restructured or a regulated electricity market. This would guarantee that the type of electricity market in which a coal plant is operating was pre-determined and completely exogenous to firms' environmental compliance decisions. Although this controlled experiment did not occur, three factors make it possible to causally relate differences in economic regulation to differences in compliance choices.

First, the timing of the NBP and electricity industry restructuring was such that a state's restructuring status was completely pre-determined. All 19 states that were ultimately included in the NBP held hearings to consider restructuring their respective electricity industries between 1994 and 1998. By 1999, restructuring bills had been passed in 12 of these states and D.C. By 2000, the remaining 7 states had all officially resolved not to move forward with electricity restructuring (EIA).[27] Consequently, when the courts upheld the NBP and the terms of environmental compliance were finally established, plant managers knew what type of electricity market they would be operating in.

Second, the factors that determined a state's restructuring decision are independent of the

[26]In the case of regulated plants, it is most common to assume that managers maximize profits subject to regulatory constraints (Averch and Johnson, 1962; Bohi and Burtraw, 1992). However, several alternative management objectives have been suggested, including maximizing returns on investment, maximizing output, maximizing revenues and maximizing reliability of supply (Bailey and Malone, 1970).

[27]Of the 19 states that are affected by the NOx SIP Call, 12 have restructured their electricity industries: CT, DE, IL, MA, MD, MI, NJ, NY, OH, PA, RI and VA. The remaining 7 chose not to go forward with restructuring: AL, IN, KY, NC, SC, TN, WV.

factors that determine compliance costs at coal-fired generating units. Most states that decided against restructuring did so because electricity rates were relatively low to begin with (Bushnell and Wolfram, 2005; Van Doren and Taylor 2004).[28] Other authors have argued that the availability of profitable nearby export markets also increased the probability that a state would pass restructuring legislation (Ando and Palmer,1998). Finally, the California electricity crisis was enough to dissuade any states who had yet to pass restructuring legislation as to whether restructuring would deliver a net gain (politically or otherwise). Momentum behind restructuring fell flat after the California electricity crisis in 2000.

Third, there is significant overlap in the distribution of the variables that determine compliance costs. Coal plants serving restructured markets are extremely similar to those serving regulated markets. Empirical analysis presented in the following section demonstrates these similarities.

III. A First Look at the Data

3.1. Data description

The data set includes the 702 coal-fired generating units that are regulated under the NBP. Of these, 322 are classified as "regulated" for the purpose of this analysis.[29] The results presented here are generated using data from 632 units.[30]

I do not directly observe the variable compliance costs and fixed capital costs or the post-retrofit emissions rates that plant managers anticipated when making their decisions. I can, however, generate unit-specific engineering estimates of these variables using detailed unit-level and plant-level data. In the late 1990s, to help generators prepare to comply with market-based NOx regulations, the Electric Power Research Institute[31] developed software to generate cost

[28] Low rates were a consequence of having access to cheap hydro and coal generation, limited investment in nuclear power, or fewer long-term fixed price contracts with independent power producers that had been encouraged under the 1978 Public Utility Regulatory Policy Act.

[29] Regulated plants include those subject to PUC regulation in states that have chosen not to restructure their electricity industries, and any state or municipally owned and operated facilities in restructured markets.

[30] Compliance costs for the remaining 70 coal fired units cannot be generated due to data limitations. These units appear on states' lists of coal-fired units in the NOx SIP Call, but appear only sporadically in EPA, EIA and Platts databases. These units appear to be significantly smaller and younger on average. The mean capacity is 22 MW compared to the sample average capacity of 252 MW (only 22 of the excluded units reporting). The mean age is 14 years, compared to a sample average of 36 years (only 4 of the excluded units reporting).

[31] The Electric Power Research Institute (EPRI) is an organization that was created and is funded by public and private electric utilities to conduct electricity industry relevant R&D.

estimates for all major NOx control options available to coal-fired boilers, conditional on unit and plant level characteristics. The software has been used not only by plant managers, but also by regulators to evaluate proposed compliance costs for the utilities they regulate (Himes, 2004; Musatti, 2004; Srivastava, 2004). I use this software to estimate capital and variable compliance costs at the unit level (EPRI, 1999b).

Cost estimation requires detailed data on over 80 unit and plant level operating characteristics (such as boiler dimensions, pre-retrofit emissions rates, plant operating costs, etc.). Together with these data inputs, the software can be used to first identify which NOx control technologies are compatible with which boilers, and then to generate boiler-specific variable costs and fixed cost estimates for each viable compliance option. Post-retrofit emissions rates are estimated using the EPRI software, together with EPA's Integrated Planning Model (US EPA 2003). Appendix B describes these data in detail.

3.2. Summary Statistics

Figures 2a and 2b summarize the observed compliance choices for units in restructured and regulated electricity markets in terms of MW of installed capacity (87,828 MW in regulated markets and 88,370 MW in restructured markets). More specifically, the figures summarizes the NOx control technology retrofits reported by these plants between 2000 and 2004. A significantly larger proportion of the coal capacity in unrestructured markets has been retrofit with SCR (the control option that is the most capital intensive and delivers the most significant emissions reductions). Conversely, in restructured markets, a greater proportion of capacity has either not been retrofitted, or has been retrofitted with controls that can achieve only moderate emissions reductions (such as combustion modifications or SNCR). These data are consistent with, but not proof of, the hypothesis introduced in the previous section.

There are several reasons why we might observe differences in compliance strategy choices across electricity market types. One appealing explanation is that this permit market is efficiently coordinating investment in pollution controls such that the plants with the lowest control costs are installing control equipment, and that SCR costs happen to be relatively high in restructured markets. Put differently, it is possible that these differences can be explained by differences in unit-specific compliance costs. Another possible explanation has to do with variation in choice sets. Because units in restructured markets have historically been subject to more stringent environmental regulations prior to the NBP, differences in adoption patterns could be attributable to the fact that generators in restructured markets were more likely to have carried out retrofits prior to 2000.

Table 1 presents summary statistics for unit-level operating characteristics that significantly determine choice sets and compliance costs: nameplate capacity, plant vintage, pre-retrofit emissions rates, pre-retrofit heat rates and pre-retrofit summer capacity factor. Overall, these two groups of coal generators look extremely similar.[32] These results indicate that the unit characteristics that help determine compliance costs are distributed similarly within the two sub-populations of coal fired units.

These two groups of units are also very similar in terms of the NOx controls installed at the time the NBP was promulgated. Over 80% of capacity in both populations had some type of low NOx burners installed; 5% of capacity in restructured markets and 7% of capacity in regulated markets had adopted some form of emissions reducing combustion modifications. No SCR retrofits had taken place in regulated markets as of 2000. Only two units had installed SCR in restructured markets.[33]

Although fifteen different compliance strategies are observed in the data; the most alternatives available to any one unit is ten.[34] With the obvious exception of the "no retrofit" option, all of the observed compliance strategies chosen by plant managers involve some combination of eight different NOx control technologies.Table 2 characterizes the choice sets which vary across units depending on unit operating characteristics and pre-existing NOx controls. The size and content of choice sets do not significantly differ across market types.

Table 3 presents summary statistics for compliance costs (estimated at the unit level) for

[32] The one dimension in which these two groups do differ somewhat is the pre-retrofit emissions rate which is lower on average among units in restructured markets. This is to be expected; because of persistent air quality problems in the Northeast, these plants have historically been subject to more stringent pollution regulation.

[33] These two units are excluded from the analysis as there was no longer a compliance choice to make.

[34] These strategies are: (1) combustion modification, (2)combustion modification combined with low NOx burners, (3) (4) (5) (6) four different types of low NOx burner technologies, (7)low NOx burners combined with SCR, (8) overfire air, (9) overfire air combined with low NOx burners, (10) SCR, (11) SNCR, (12) SCR with overfire air, (13) SNCR with overfire air, (14) low NOx burners, SCR and overfire air, (15) no retrofits.

the most commonly adopted technologies. There are no significant differences in average costs across the two electricity market types.[35] Taken together, these descriptive statistics suggest that variation in compliance costs and choice sets is insufficient to explain the substantial differences in observed compliance choices across market regimes.

4. An Empirical Model of the Compliance Choice

In this section, I develop an empirical model of a plant manager's choice between mutually exclusive approaches to complying with this emissions trading program. The purpose of specifying the model is twofold. First, it provides a framework to test whether economic regulation has affected the environmental compliance choice. Second, the model provides a means to evaluate how these plant managers would have responded to a permit market designed to reflect spatial variation in marginal damages from pollution.

This analysis focuses exclusively on the compliance choices that were made in the years leading up to the compliance deadline (2000-2004).[36] Because it is difficult to identify the precise point in this four year period at which this decision was made, these compliance choices are modeled as static decisions.[37]

The manager of unit n faces a choice among J_n compliance strategy alternatives (indexed by j, $j = 1...J_n$). Plant managers are assumed to choose the compliance strategy that minimizes the unobserved latent variable C_{nj}. The deterministic component of C_{nj} is a weighted sum of expected annual compliance costs v_{nj}, the expected capital costs K_{nj} associated with initial retrofit and

[35] Average costs are slightly higher for units in more regulated electricity markets. This is likely due to the fact that plants with higher pre-retrofit emissions rates tend to have higher retrofit costs.

[36] Past research has cautioned against trying to identify differences in the underlying propensity to adopt a new technology using choices observed over a short time period. Particularly in the case of a "lumpy", capital intensive technology, the pattern of technology diffusion across firms can be driven by differences in opportunities to adopt (Rose and Joskow, 1984). Fortunately, the NOx SIP Call eliminates temporal variation in technology adoption opportunity by design; every coal plant manager was forced to make a decision of how to comply with the program during the four years between when terms of compliance were officially established and when full compliance was required of all plants.

[37] Because of labor shortages and a limited number of tower-cranes needed to complete SCR retrofits, many plants reported delays of several years between when they made their compliance decision and when the pollution control retrofit was completed (Cichanowicz, 2004; Midwest Construction, 2005). Consequently, reported retrofit dates are a very noisy measure of when the compliance decision was actually made. There is arguably a dynamic component to the compliance strategy choice that is ignored by this specification. Plants could postpone the decision to invest in pollution controls until after the NOx SIP Call program had taken effect. However, because more pollution control equipment was installed than is needed to comply with SIP Call, the decisions analyzed here will determine regional emissions patterns to a significant extent for the foreseeable future (Natural Gas Week, 2004).

technology installation, and a constant term α_j that varies across technology types :

$$
\begin{aligned}
(1) \qquad\qquad C_{nj} &= \alpha_j + \beta_n^v v_{nj} + \beta_n^K K_{nj} + \beta_n^{KA} K_{nj} \cdot Age_{nj} + \varepsilon_{nj}, \\
where \quad v_{nj} &= (V_{nj} + \tau m_{nj}) Q_n
\end{aligned}
$$

An interaction term between capital costs and demeaned plant age is included in the model. Older plants can be expected to weigh capital costs more heavily as they have less time to recover these costs. The variable cost (per kWh) of operating the control technology is V_{nj}. The variable costs associated with offsetting emissions with permits is equal to the permit price τ multiplied by the post-retrofit emissions rate m_{nj}.[38] Expected average annual compliance costs are obtained by multiplying estimated per kWh variable costs by expected seasonal production Q_n.

Expected seasonal electricity production at a unit (Q_n) is assumed to be independent of the compliance strategy being evaluated. Anecdotal evidence suggests that managers used past summer production levels to estimate future production, regardless of the compliance choice being evaluated (EPRI, 1999a). I adopt this approach and use the historical average of a unit's past summer production levels ($\bar{Q}_n$) to proxy for expected ozone season production. Empirical support for this assumption is presented in section 6.3.

It is likely that the compliance choice characteristics that are relevant to the compliance decision are not limited to observable cost characteristics. Technology constants α_j capture unobserved, intrinsic technology preferences or biases such as widely held perceptions regarding the reliability of a particular type of NOx control technology. A stochastic component ε_{nj} is included in the model to capture the idiosyncratic effect of unobserved factors.

This reduced form model has just enough structure to capture the differences in responsiveness to capital costs and variable costs across units, and across electricity market types more generally. It is straightforward to map the parameters in this model to the parameters in the economic model specified in Appendix A. This allows for a more structured interpretation of the estimated

[38] The unit-specific, compliance strategy-specific estimates of K_{ni} and V_{ni} are generated using the EPRI cost estimation software described in section 4.1. Emissions rates (which also vary across units and control technologies) are estimated using the software and accompanying documentation and EPA's IPM model (US EPA 1998d), in addition to other sources in the technical literature which are discussed in the data appendix.

coefficients; the cost coefficients can be viewed as functions of a plant's cost of capital, cost recovery parameters, and the scale parameter of the extreme value distribution. However, it is not clear that cost minimization is the most accurate way to characterize the objective functions of all plant managers. This model is sufficiently general to accommodate a variety of possible objectives.

A. The Conditional Logit Model

I first estimate a conditional logit (CL) model of the compliance decision. Conditional on observed unit characteristics, coefficients are not permitted to vary across units. The ε_{nj} are assumed to be iid extreme value and independent of the covariates in the model.[39]

Let y_n be a scalar indicating the observed compliance choice, $y_n \in \{1, ...J_n\}$.. The closed form expression for the probability (conditional on the vector of coefficients β and the matrix of covariates X_n) that the n^{th} unit will choose compliance strategy i is:

$$P(y_n = i|X_n, \beta) = \frac{e^{-\beta' X_{ni}}}{\sum_{j=1}^{J_n} e^{-\beta' X_{nj}}}. \quad (2)$$

This conditional choice probability is derived in Appendix C.

B. The Random Coefficient Logit Model

The CL model, however elegant, is not the best choice for this application. First, this model does not account for random variation in tastes or response parameters; conditional on observed plant characteristics, the coefficients in the model are not allowed to vary across choice situations. There are likely to be factors affecting how plant managers weigh compliance costs in their decision-making that we do not observe. Examples include variation in plant's costs of capital, managerial attitudes towards risk, contractual arrangements, and subtle variations in PUC cost recovery rules. To the extent that variation in unobserved determinants of the compliance choice is significant, errors will be correlated and CL coefficient estimates will be biased.

[39] This stochastic term is subtracted from (versus added to) the deterministic component of costs in order to simplify the derivation of choice probabilities implied by this model (see Appendix 3). These choice probabilities are very similar to the standard logit choice probabilities derived under assumptions of random utility maximization (McFadden, 1973).

The second limitation has to do with the panel structure of data used to estimate the model. While I only observe one compliance choice for each coal-fired boiler or "unit", an electricity generating facility or "plant" can consist of several physically independent generating units, each comprising of a boiler (or boilers) and a generator. Some plants only have one boiler, but there can be as many as ten boilers at a given plant. The 632 boilers in the sample represent 221 power plants owned by 86 different companies or public agencies. It seems reasonable to assume that the same plant manager made compliance decisions for all boilers at a given plant. It is also possible that compliance decisions could be correlated across facilities owned by the same parent company. The CL model cannot accommodate this correlation across choice situations associated with the same decision maker.

The random-coefficient logit (RCL) model, a generalization of the CL model, does a better job of accommodating unobserved response heterogeneity and relaxes the troublesome iid error structure assumption. This specification allows one or more of the model parameters to vary randomly across decision makers. I assume that the variable cost coefficient (β^v) and the capital cost coefficient (β^K) are distributed in the population according to a bivariate normal distribution, thereby accommodating any unobserved heterogeneity in responses to changes in compliance costs.

I maintain the assumption that the unobserved stochastic term ε_{nj} is iid extreme value and independent of β and X_{nj}. To accommodate the panel nature of the data, the (unobserved) β vectors are allowed to vary across managers according to the density $f(\beta|b, \Omega)$, but are assumed to be constant over the choices made by a manager.[40] Thus, the coefficient vector for each manager (indexed by m) can be expressed as the sum of the vector of coefficient means b and a manager-specific vector of deviations η_m. Because the η_m are assumed to be equal across choices made by the same manager (at the same plant), the unobserved component of anticipated costs is correlated within a plant. This does not imply that the errors corresponding to all choices faced by a single manager are perfectly correlated; the extreme value error term still enters independently for each choice.

[40] Alternatively, beta vectors could be held constant across all units, and across all plants owned by the same parent company. Interviews with industry representatives indicate that it is sometimes the case that environmental compliance decisions are made or influenced by the parent company (Whiteman, 2005). A model where cost coefficients are allowed to vary across parent companies, but not across plants, is also estimated.

Conditional on β_m, the probability that a manager of a plant comprised of T_m units makes the observed Y_m compliance choices is:

$$(3) \qquad P(Y_m = \boldsymbol{i}|X_m, \beta_m) = \prod_{t=1}^{T_m} \frac{e^{-\beta'_m X_{mit}}}{\sum_{j=1}^{J_{mt}} e^{-\beta'_m X_{mjt}}},$$

where $\boldsymbol{i}$ is a $T_m * 1$ dimensional vector denoting the set of observed choices made by manager m. Here, the n subscript denoting the unit has been replaced by a unique mt pair. Unconditional choice probabilities $P(Y_m = \boldsymbol{i})$ are derived by the integrating conditional choice probabilities over the assumed bivariate normal distribution of the unobserved random parameters.

The unknown vector of coefficient means b and covariance matrix Ω describe the distribution of the β_m in the population.[41] Parameter estimates are those that maximize the following log likelihood function:

$$(4) \qquad LL(b, \Omega) = \sum_{m=1}^{M} \ln \int_{-\infty}^{\infty} \prod_{t=1}^{T_m} \frac{e^{-\beta'_m X_{mit}}}{\sum_{j=1}^{J_{mt}} e^{-\beta'_m X_{mjt}}} f(\beta|b, \Omega) d\beta.$$

Unconditional probabilities are approximated numerically using simulation methods. The RCL estimates are those that maximize the simulated likelihood function. For each decision maker, 1000 two-dimensional vectors of independent standard normal random variables are drawn. To simulate a random draw from the bivariate normal density $f(\beta|b, \Omega)$, each vector of standard normals is multiplied by the cholesky factor L of the covariance matrix and the resulting product is added to the vector b. To increase the accuracy of the simulation, pseudo-random Halton draws are used (Bhat 1998; Train, 2001).[42] The value of the integrand [3] is calculated for each decision

[41] The model is parameterized in terms of the Cholesky factor L of the covariance matrix Ω, so as to allow the two random cost coefficients to be correlated. Because the covariance matrix is positive definite, it can be expressed as the product of the lower triangular matrix L and its transpose.

[42] Researchers have found that using Halton draws (versus random draws) provide more uniform coverage over the domain of the integration spaceand results in more accurate computation of probabilities for a given number of draws. Bhat(2003) finds that 125 Halton draws produces more accurate estimates than 2000 random draws.

maker, for each draw. The results are averaged across draws. The maxlik algorithm in Gauss is used to find estimates of the parameters in b and L that maximize the simulated likelihood of the observed compliance choices.[43] To estimate standard errors, the robust asymptotic covariance matrix estimator is used (Mc Fadden and Train, 2000).

C. Manager Specific Parameters

The RCL estimates of b and Ω provide information about how the capital and variable cost coefficients are distributed in the population, but tell us nothing about where one manager lies in the distribution relative to other managers. Recent work demonstrates how simulated maximum likelihood estimates of random-coefficient, discrete choice models can be combined with information about observed choices in order to make inferences about where in the population distribution a particular agent most likely lies (Allenby and Rossi, 1999; Revelt and Train, 2000; Train, 2003).[44]

Following Train (2003), let the density describing the distribution of β in the population of managers be denoted $g(\beta|b,\Omega)$. The probability of observing the m^{th} manager making the choice he does when faced with the compliance decision described by the matrix of covariates X_m is given by [4]. This probability is conditional on information we cannot observe (β_m). The marginal probability of observing this outcome is $P(Y_m|X_m,b,\Omega) = P(Y_m = \boldsymbol{i}|X_m,\beta)g(\beta|b,\Omega)$. Let $h(\beta|\boldsymbol{i},X_m,b,\Omega)$ denote the distribution of β_m in the sub-population of plant managers who, when faced with the compliance choice set described by X_m, would choose the series of strategies denoted $\boldsymbol{i}$. Applying Bayes rule, this manager specific, conditional density of β_m can be expressed:

$$(5) \qquad h(\beta|\boldsymbol{i},X_m,b,\Omega) = \frac{P(Y_m = \boldsymbol{i}|X_{m,}\beta)g(\beta|b,\Omega)}{P(Y_m = \boldsymbol{i}|X_m,b,\Omega)}.$$

These conditional distributions are implied by the simulated maximum likelihood estimates of the population distribution parameters and the choices we observe. To illustrate this more

[43] Gauss code is based on that developed by Train, Revelt and Ruud (1999).

[44] Alternatively, a finite mixture logit (FML) model could have been estimated in order to obtain information about where in the larger population distribution a particular type of manager lies.However, a demonstrated limitation of these models is that they often cannot adequately capture all of the heterogeneity in the data (Allenby and Rossi, 1999; Rossi et al. 1996).

explicitly, [5] can be reformulated as:

$$h(\beta|\boldsymbol{i}, X_m, b, \Omega) = \frac{\prod_{t=1}^{T_m} \frac{e^{-\beta'_m X_{mit}}}{\sum_{j=1}^{J_{mt}} e^{-\beta'_m X_{mjt}}} g(\beta|b,\Omega)}{\int_{-\infty}^{\infty} \prod_{t=1}^{T_m} \frac{e^{-\beta'_m X_{mit}}}{\sum_{j=1}^{J_{mt}} e^{-\beta'_m X_{mjt}}} g(\beta|b,\Omega) d\beta}. \tag{6}$$

These conditional distributions can be used to derive conditional expectations of functions of β. For example, the expected probability that alternative i will be chosen by the m^{th} manager in a counterfactual choice situation denoted $T+1$ can be expressed as:

$$E[P(y_{m,T+1} = i|Y_m, X_m, b, \Omega)] = \frac{\int_{-\infty}^{\infty} \prod_{t=1}^{T_m+1} \frac{e^{-\beta'_m X_{mit}}}{\sum_{j=1}^{J_{mt}} e^{-\beta'_m X_{mjt}}} g(\beta|b,\Omega)}{\int_{-\infty}^{\infty} \prod_{t=1}^{T_m} \frac{e^{-\beta'_m X_{mit}}}{\sum_{j=1}^{J_{mt}} e^{-\beta'_m X_{mjt}}} g(\beta|b,\Omega) d\beta}, \tag{7}$$

A simulated approximation to this expectation is obtained by first drawing from the estimated population distribution $g(\beta|b, \Omega)$ and then simulating conditional values of the counterfactual choice probability for each draw.[45]

5. Estimation

Tests of the hypothesis introduced in Section 3 can be formulated as a test of whether the random parameter estimates differ significantly across electricity market types. There are two possible approaches to comparing coefficient estimates across groups. First, a single model that includes interactions between the coefficients of interest and a dummy variable indicating group membership can be estimated using pooled data. A second approach involves estimating the model separately for each group.

[45] This approach involves integrating over the estimated distribution of the random coefficients in the population; this formulation accounts for sampling and simulation error in estimates of b and Ω. Integrals are simulated in the same way as for the unconditional RCL choice probabilities.

The first approach implicitly assumes that the variance of the disturbance term is equal across groups (Allison, 1999). Because the extreme value error term is likely capturing different unobserved variables in the restructured and regulated cases, this assumption is unlikely to be met.[46] Consequently, the results from estimating a single model using pooled data are underemphasized.

The advantage of the second approach is that coefficient estimates and standard errors are consistent within each group. In order to identify the logit model, all coefficients have been scaled by the scale parameter of the extreme value distribution. When the model is estimated separately using data from restructured and regulated markets, direct comparisons of coefficients across the two groups are confounded by this identification assumption. Within a model, however, tests of the significance of a given coefficient are valid; the ratio of the coefficient and the variance of the unobserved stochastic term will only be zero if the coefficient is zero. Consequently, such comparisons can be informative if the pattern of coefficient significance varies across groups.

5.1. Conditional logit model results

The first two columns of Table 4 report estimates for the more restrictive CL specification in which coefficient values are not permitted to vary across plant managers. In both the restructured and regulated cases, a nested likelihood ratio test of this specification against a benchmark specification that includes only technology specific constants indicates that including variable and capital cost variables significantly improves the fit of the model.[47]

All of the technology type constants are negative and significant at the 1 percent level, regardless of whether the CL model is estimated using data from regulated or restructured markets.[48] One interpretation of this result is that, relative to the baseline option of no control technology

[46] Monte Carlo experiments have illustrated that the most likely outcome of estimating a single equation with interaction terms when the residual variances differ across groups is that the slope coefficients will be found not to differ even if they actually do, but it is also possible to find an effect when no effect exists (Hoetker, 2003).

[47] The fit of the nested (or more restrictive) model can be evaluated using a chi-square statistic. This test statistic is calculated by taking twice the absolute difference in the log likelihoods for the two models. If significant, (degrees of freedom are equal to the difference in the number of parameters between the two models), the nested model should be rejected (Bhat, 1998). The test statistics reported in the last row of Table 3 are larger than the χ^2 statistic with 3 degrees of freedom and a p-value of 0.001.

[48] I include only three technology fixed effects for the three major categories of NOx controls: Post-combustions pollution control technologies (SNCR and SCR), Combustion Modifications (CM) and Low NOx Burner (LNB) technologies. Although cost estimates and emissions reduction estimates were generated for sub-classes of these categories (for example, there are four different types of low NOx burners in the data), including a more complete set of technology fixed effects did not improve the fit of the model.

retrofit, managers were biased against retrofits in general (controlling for costs).

The coefficient on variable compliance costs is statistically significant at the 1 percent level and has the expected negative sign in both the regulated and restructured electricity market cases. These results indicate that expected variable compliance costs are an important factor affecting the plant's compliance choice. When the model is estimated using data from restructured electricity markets, the coefficient on capital costs is statistically significant and has the expected negative sign. An increase in the capital cost of a compliance option decreases the probability that the option will be chosen by a plant in a restructured electricity market. However, when the model is estimated using data from regulated electricity markets, the coefficient estimate is positive and is not statistically significantly different from zero, suggesting that capital costs might not be a significant factor in the compliance decisions at regulated plants.[49]

5.2. Random Coefficient Logit Results

Results from estimating the RCL model are presented in the third and fourth columns of Table 4. Estimated standard deviations of the two random coefficients are statistically significant. The results of a nested likelihood ratio test imply that, in both the restructured and regulated cases, allowing for response heterogeneity significantly improves the fit of the model. These results suggest that cost coefficients vary significantly across managers in regulated and restructured markets.[50]

When the model is estimated using data from restructured markets, the means of both the capital and variable compliance cost coefficients are negative and significant at the 1 percent level. The estimated standard deviations are also large in absolute value and statistically significant, indicating that there is unobserved variation in responsiveness to changes in compliance costs.[51]

[49] A single model was estimated using pooled data. Interactions between cost variables and a dummy variable indicating a restructured electricity market are included in this model. Whereas the coefficient on the uninteracted capital cost variable is not statistically significant, the estimated coefficient on the interaction between capital costs and the restructured market indicator is statistically significant and has the expected negative sign. These results are consistent with the results in Table 4.

[50] These RCL estimates are robust to various optimization routines and variation in the number of pseudo-random draws used in the simulations.

[51] There are several possible explanations for this variation, including variation in costs of capital and variation in managers' risk aversion. In an effort to attribute some of this variation to observable plant characteristics (such as plant size and whether or not the plant had been divested), other interactions were also tested, but none improved the fit of the model.

The negative and significant coefficient values on the capital cost/age interaction term indicates that older plants weighed capital costs more negatively in their compliance decision, presumably due to shorter investment time horizons.

Different results are obtained when the model is estimated using data from regulated markets. The point estimate for the capital cost coefficient is substantially smaller than the point estimate obtained using data from restructured markets, and is not statistically significant at the 1 percent or 5 percent level. The standard deviation of this coefficient is significant, suggesting that there is unobserved heterogeneity in how responsive managers are to variation in capital costs. The capital cost/age interaction term is significant and has the expected negative sign. Among older regulated plants, the capital cost coefficient is statistically significant, possibly because regulators are unlikely to approve a major capital investment in pollution control equipment if the plant is very old and expected to retire soon. The variable cost coefficient is also statistically significant and negative when the model is estimated using data from regulated electricity markets.

The RCL estimates of the moments of the distribution of β in the population are combined with the observed choices in order to derive the parameters of manager specific conditional distributions. The population parameter estimates $\hat{b}$ and $\hat{\Omega}$ are substituted into [6] and the first and second moments of these conditional distributions are calculated (using the same matrix of Halton draws that were used to estimate [5]). Table 5 presents the summary statistics for the estimated moments of these 221 manager-specific distributions. If the model is correctly specified, the average of the means of the manager specific conditional distributions (the $\bar{\beta}_m s$) should be very close to the estimated population means. These results offer no evidence to suggest that the normality assumptions are inappropriate. The standard deviations of the conditional means are significantly larger than zero, suggesting that variation in the conditional means captures a significant portion of the total estimated variation (Revelt and Train, 2000).

The elasticities implied by the model estimates provide a more intuitive characterization of the responsiveness of compliance decisions to changes in compliance costs. Table 6 presents average elasticities with respect to both own capital costs and own average ozone season variable compliance costs for the most commonly observed compliance choices. Elasticities for each choice situation are calculated using point estimates of the means of the corresponding manager-specific conditional distributions. These summary statistics indicate that choice probabilities in restructured markets are, on average, more sensitive to changes in compliance costs in general, and capital costs in particular. We should be most interested in how changes in costs affect the probability of adopting the cleanest and most capital intensive technology: SCR. The model predicts that a one percent increase in the capital cost of an SCR retrofit, holding all else equal, will result in an average decrease of 5.7 percent in the probability that SCR will be chosen among units in restructured electricity market. This average decrease is 1.3 percent in regulated markets. The corresponding variable cost elasticities are 1.8 and 1.3, respectively. The standard deviations of these elasticity estimates are reported in parentheses.

One way to get around the scaling problem that confounds direct comparisons of these coefficients across groups is to compare ratios of coefficient estimates. The ratio $\beta^K : \beta^v$ can be interpreted, under certain assumptions, as an estimate of the discount rate (see Appendix A). The point estimates of this ratio is 44% and 16% in restructured and regulated markets, respectively. This ratio can also be estimated at the unit level using manager-specific coefficient estimates. When the ratio $(\beta_m^K + \beta_m^{KA} \cdot A_{nt}) : \beta^v$ is estimated for each unit, two distributions of ratio estimates are generated. The mean and standard deviation of these distributions are 33.7% ($\sigma = 120\%$) and 7.7% ($\sigma = 24.2\%$).in restructured and regulated markets, respectively.[52]

These results suggest that, on average, managers in regulated electricity markets were willing to tolerate higher up-front costs in order to lower their variable compliance costs, as compared to managers in restructured electricity markets. Making formal statistical inferences about the difference between these two ratio estimates requires standard error estimates. Unfortunately, more standard approaches to estimating the variance of a function of random variables (such

[52] Researchers have in the past made simplifying but restrictive assumptions in order to circumvent problems associated with estimating the parameters of the distribution of a ratio of random parameters. One common approach involves assuming that the coefficient in the denominator is fixed (Hensher et al, 2004; Layton and Brown, 2000). Sonnier et al. (2005) show that constraining the coefficient in the denominator to be fixed in order to get a ratio that is normally distributed results in an overestimate of the variance of the ratio, even when the true variance is small. Other reseachers have reparameterized the RCL model so as to identify the ratio directly. Rather than set the scale parameter to one, one of the coefficients in the model is restricted to equal one (Train and Weeks, 2004; Sonnier et al. 2005). This approach is inappropriate for this application, where the capital cost and variable cost coefficients are likely to differ across models.

as using the delta method or a bootstrap) are inappropriate here.[53] Standard deviations of the manager-specific, technology-specific elasticity estimates are reported.

5.3. Further Robustness Tests

Company versus plant manager specific coefficients

Many of the facilities analyzed here are owned by a common parent company. If the environmental compliance decision was made at the company (versus manager) level, a specification that allows for correlation in choices made across facilities owned by the same parent company would be more appropriate. An RCL model that restricts the cost coefficients to be equal across units owned by the same parent company was also estimated.

Table 7 reports the estimation results. Patterns of coefficient significance are robust to specification choice. Whereas the null hypothesis that the capital cost coefficient equals zero can easily be rejected in the restructured market case, it cannot be rejected in the regulated market case. Similar to the results generated under less restrictive assumptions of manager-specific coefficients, estimating the parameters of company-specific distributions lend support to the assumption of a bivariate normal distribution for the random parameters. The point estimates of the average ratio $\beta_p^K : \beta_p^v$, (where p denotes parent company) are 0.30 and 0.10 in restructured and regulated markets, respectively.[54]

Alternative specifications

Section 3 offered several reasons why plant managers (or owners) in regulated markets might be more likely to adopt more capital intensive compliance options, including an Averch and Johnson effect, lower costs of capital, and less uncertainty about capital cost recovery. In the interest of trying to tease apart the relative importance of these factors, several alternate specifications were tried. For example, in restructured electricity markets, cost variables were interacted with a

[53]The delta method is often used to estimate the standard error of ratio statistics, based on a first order Taylor series expansion of the ratio centered at the mean of b. The delta method cannot be used here because the variance of $\beta^K : \beta^v$ is not well-defined. The same problem arises if a bootstrap is used to estimate the standard errors of these ratios. The support of the estimated distribution of β^v for both restructured and regulated electricity markets overlaps zero. With enough samples, the bootstrap eventually generates estimates of β^v that are arbitrarily close to zero, implying infinitely large estimates of the ratio.

[54]Ideally, a formal statistical test would be carried out to determine which of these two specifications is most consistent with the data. Classical inference based on log-likelihood ratio statistics is invalid because these are non-nested models. A formal test of these non-nested hypotheses is beyond the scope of this paper (see Vuong, 1989).

dummy indicating that the plant had been divested. Divested (or recently purchased) plants would have high debt:equity ratios and higher costs of capital. In the regulated model, cost variables were interacted with dummy variables indicating whether the unit was a government owned or investor owned plant. None of these interaction terms significantly improved the fit of the model.

Testing the exogeneity of Q_n

A final test pertains to how plant managers formed their expectations about future production. I have assumed that production expectations are independent of the compliance alternative being evaluated. The average of a unit's past summer production levels in the years preceding the compliance decision $\bar{Q}_n$ is used to proxy for expected ozone season production. Because coal generation tends to serve load on an around-the-clock basis, the capacity factors of most coal plants are unlikely to be significantly affected by a compliance-related change in variable operating costs.[23] However, if $\bar{Q}_n$ consistently under (or over) estimates what managers actually expected, the variable operating cost measures will be biased.

It is impossible to know whether all plant managers used $\bar{Q}_n$ to approximate Q_n in their decision making.[55] However, unit level production data from the first ozone season can be used to assess how well $\bar{Q}_n$ predicts the electricity production we do observe.[56] The following equation is estimated:

$$Q^*_{n,04} = \theta_0 \bar{Q}_n + \theta_j \sum_{j=1}^{J_n} D_{jn} \cdot \bar{Q}_n + u_n, \tag{8}$$

where $Q_{n,04}$ is the observed production at unit n during the 2004 ozone season, D_{jn} is an indicator for whether unit n adopted pollution control technology j, and u_n is a random error term. A robust covariance matrix estimator that accounts for within plant correlation in the error terms is used.[57] If unit-level production was significantly affected by firms' compliance decisions, one or more of the θ_j will be statistically significant. A positive (negative) θ_j indicates that, on average, firms choosing compliance strategy j increased (decreased) their production relative to those units

[55] Anecdotal evidence indicates that managers used past summer production levels to estimate future production, regardless of the compliance choice being evaluated (EPRI, 1999a).

[56] The first ozone season in which all coal-fired units had to comply was 2004.

[57] There are several reasons why the error terms might be correlated across units in the same facility. For example, an facility-wide outtage would affect the production of all units at a plant.

who chose to rely entirely on the permit market for compliance.

Results are reported in Table 8. The coefficient on $\bar{Q}_n$ is 1.03 when the model is estimated using data from the regulated markets and very precisely estimated, whereas none of the interaction terms are significant. This implies that unit level production, on average, increased slightly in regulated markets once the NBP took effect, but was not significantly affected by the compliance strategy chosen. When the model is estimated using data from restructured markets, the coefficient on $\bar{Q}_n$ is 1, also with a small standard error. Only the SCR interaction term is positive and significant at the five percent level. This is an interesting, but not surprising result. In restructured markets, units installing SCR slightly increased their ozone season production on average, where as production levels at all other plants were generally unchanged.

These results are supportive of the model assumptions in regulated markets. If managers correctly anticipated how compliance decisions would affect future production, they used past ozone season production as a proxy for future production in their evaluation of all compliance options. In restructured markets, managers who correctly anticipated that adopting SCR (and possibly SNCR) could result in increased production (by a quantity denoted by ΔQ_n) would have changed their production expectations accordingly. This would increase annual compliance costs associated with SCR by $\Delta v_{n\ SCR} = (V_{n\ SCR} + \tau \cdot m_{n\ SCR})\Delta Q_n$.[58] Per kWh compliance costs are relatively low for SCR (see Figure 1), so $\Delta v_{n\ SCR}$ should be small. Because it is hard to know whether managers correctly anticipated this increase, and because the increase is likely to be small, the same assumptions regarding expected production are maintained for all units, for all compliance strategies.

5.4. Summary of Estimation Results

Because of the identification assumptions underlying the logit model and the difficulties associated with estimating the variance of a ratio of two random variables, there is no completely satisfying

[58] In fact, this increase in per kWh compliance costs would potentially be offset by increased revenues. Under the assumption that expected production is independent of the compliance choice, revenues from the sale of electricity do not vary across compliance alternatives and therefore drop out of the discrete choice model. If expected production is higher conditional on adopting SCR, revenues will increase by an amount equal to $\sum_{t_{n\ SCR}=1}^{T_{nSCR}} q_{nt_{nSCR}} P_{nt_{nSCR}}$, where t_{nSCR} indexes the additional hours in which the n^{th} unit would operate if it installed SCR, and P_{nt} is the electricity price the n^{th} unit expects to receive in hour t.

way to formally demonstrate that the relative magnitude of the means of the two cost coefficient distributions differs across electricity market types. However, the empirical evidence strongly suggests that the negative coefficient on capital costs is substantially larger in absolute value when the model is estimated using data from restructured electricity markets. Whereas we can easily reject the null hypothesis that the capital cost coefficient is greater than or equal to zero in the restructured market case, we fail to reject this hypothesis when the model is estimated using data from regulated electricity markets. When the ratio of the variable and capital cost coefficient estimates are compared (hereby eliminating the scale parameter that confounds direct comparisons of coefficients across market types), we find further support for the hypothesis that plants in restructured electricity markets weigh capital costs more heavily in their compliance decisions.

6. Implications of the Results

6.1. Implications for technical efficiency

Estimation results suggest that it is not always the plants with the lowest abatement costs that install pollution control technologies. To assess the magnitude of technical inefficiency, engineering cost estimates associated with observed compliance choices are compared with a stylized, compliance cost minimizing counterfactual.

A deterministic model that simulates efficient pollution permit market clearing is specified. The model is used to identify the set of compliance strategies that minimizes the sum of estimated hardware and operating costs subject to an exogenously set cap. The cap is set equal to the (undiscounted) emissions reductions associated with observed compliance choices. The model assumes that each unit chooses the compliance option that minimizes the present value of discounted compliance costs. To determine the relevant investment time horizon, I assume all units retire at 65 years. I use the financial parameters typically assumed by federal and state regulatory agencies when analyzing industry pollution regulation (i.e. IPM model assumptions) to discount future costs (EPA, 2003).

Table 9 reports some results from this exercise. The estimated net present value (NPV) of discounted compliance costs associated with observed choices is $9.3 B, whereas the estimated NPV of discounted costs associated with the set of choices that deliver the same emissions reductions at minimum cost is \$6.7 B. The deterministic model predicts that investment in pollution control will be divided approximately equally across electricity market regimes.[59] Under cost minimization, however, 61% of investment occurs in regulated markets.

Note that the costs associated with observed choices exceed cost minimizing levels in both market regimes. The deterministic model is overly simplistic in assuming that all firms use the same discount rates, costs of capital, etc. when making their compliance decisions. In restructured markets in particular, this was certainly not the case.[60] What this exercise does illustrate, however, is that restructured markets as a whole were much closer to the stylized, cost-minimizing level of investment, as compared to regulated producers.

6.2. Implications for Permit Market Design

Ozone non-attainment problems are significantly more severe in states that have restructured electricity markets, largely because of differences in levels of industrial activity, population densities, and meteorological conditions. Consequently, the health benefits from reducing NOx pollution are significantly greater in these states.

Consider the health effects of choosing to install selective catalytic reduction (SCR) technology (the most capital intensive NOx control option) at a unit in a regulated electricity market versus a unit in a restructured electricity market. An average unit in the sample emitted 15 tons of NOx per day in 1999; retrofitting a *single unit* with SCR technology results in daily NOx reductions of 12 tons on average. A recent study finds that shifting 11 tons of NOx emissions per day from a relatively "low damage" location (North Carolina, a state that has not restructured its electricity market) to a "high damage" area (Maryland, a state that restructured its electricity industry) over a ten day period results in the loss of approximately one human life (Mauzerall et al., 2005).[61] If there were two technically identical plants located in Maryland and North Carolina, respectively, we would much rather see the investment in SCR occur at the plant in Maryland. However, results

[59] This is not surprising; recall that units are divided, and technology costs are distributed, very similarly across market regimes (see Tables 1 and 2).

[60] For example, firms that had recently divested generation assets could finance investments in pollution control equipment relatively more easily than firms who had recently purchased a divested plant.

[61] Recent epidemiological studies indicate that health impacts increase linearly with increasing ozone concentrations (US EPA, 2003; Steib et al., 2003, as cited in Mauzerall et al., 2005).

presented in the previous section indicate that if these two plants faced the same choice set, it is more likely that the investment would occur in North Carolina.

Like all major CAT programs in the United States, the NBP is emissions-based. The regulatory constraint is defined in terms of pounds of pollution; a permit is worth a pound of emissions, regardless of where the pound is emitted. Because the permit market fails to reflect spatial variation in benefits from reducing NOx emissions, there will likely be insufficient incentives for efficient levels of investment in the regions where investment in pollution controls will deliver the greatest benefits. Because air quality problems are more severe in states that have restructured their electricity markets, the allocative inefficiencies associated with emissions-based trading of a non-uniformly mixed pollutant are exacerbated by the economic regulation effects discussed in the previous section.[62]

Whereas environmental regulators have no control over electricity market regulation, they do have control over how pollution permit markets are designed. An alternative approach to designing permit markets involves setting a cap on total damages and establishing trading ratios that determine the terms of interregional permit trading.[63] To set up such a system, the marginal damages resulting from increased NOx emissions in different regions of the regulated area must be estimated. The trading ratio R corresponding to a particular region is set equal to the estimated damages for that region divided by the damages in a designated numeraire region. These regions can be as small as the available data on marginal damages allows. In the extreme case, ratios would be set at the facility level. Under emissions-based trading, $R_n = 1 \; \forall \; n$. The introduction of trading ratios that reflect spatial variation in marginal damages increases the marginal cost of polluting in areas where pollution does the most damage, thereby increasing the incentives to install pollution

[62] It is worth noting that it need not have happened this way. If marginal damages from pollution were lower in states with restructured electricity industries, the two effects would work in opposing directions.

[63] It should be emphasized that policy makers did think about incorporating trading ratios into the design of the NBP. The EPA received over 50 responses when, during the planning stages of the NOx SIP Call, it solicited comments on whether the program should impose restrictions on interregional trading in order to reflect the significant differential effects of NOx emissions across states(FR 63(90): 25902). Most commentors supported unrestricted trading and expressed concerns that "discounts or other adjustments or restrictions would unnecessarily complicate the trading program, and therefore reduce its effectiveness" (FR 63(207): 57460). A deterministic simulation exercise similar to the one discussed in the previous section was carried out. Cost-minimization was assumed and interstate variation in electricity market regulation was not represented. Simulation results indicated that imposing spatial constraints on trading would not result in significant shifts in the location of emissions. Consequently, the program was designed so that emissions are traded on a one-for-one basis.

controls in relatively high damage areas. The effect of trading ratios on compliance decisions, and thus patterns of emissions, will depend on how responsive firms' compliance choices are to changes in variable compliance costs. If the bias of managers against capital intensive compliance options is sufficiently strong in high damage areas, it could be that the use of trading ratios would not have affected compliance choices.

In the interest of assessing how the use of NOx trading ratios would affect compliance decisions, we want to compare observed compliance choices with the choices that would have been made under exposure based trading. The econometric model can be used to simulate these counterfactual compliance decisions. Drawing from the manager-specific distributions of cost coefficients implied by the RCL estimates, I simulate the compliance choices that these managers most likely would have made had the NOx emissions market been designed to reflect spatial heterogeneity in marginal damages from pollution. Unlike previous studies,[64] I will find that the decision to adopt an emissions-based versus an exposure-based permit market has significantly affected the spatial distribution of permitted emissions.

6.3. Simulating Exposure-Based Trading

Defining trading ratios

Several assumptions had to be made in setting up the simulation of exposure-based NOx permit trading. The first set of assumptions pertain to how trading ratios are defined. Although there was discussion of imposing spatial constraints on permit trading during the planning stages of the NBP, a complete proposal of appropriate jurisdictional boundaries or trading ratios was never established. However, there are two papers in the literature which estimate marginal damages from incremental increases in NOx emissions in the Eastern United States that provide estimates of marginal damages that can be used to construct blunt estimates of trading ratios (Krupnick

[64] Farrell et al. (1999) consider imposing geographic constraints on NOx permit trading in the Northeast and conclude that the benefits do not justify the costs. Krupnick et al. (2000) argue that there is no clear benefit to spatially differentiated NOx trading. Finally, the EPA used the IPM model to simulate exposure based trading under the NBP (1998c). Results suggested estimated benefits did not justify the added complexity.

et al., 1998; Mauzerall et al., 2005).[65,66] Based on these papers, I consider two exposure-based trading scenarios. In both cases, one permit is required to offset a pound of pollution in low damage areas. In "high damage" areas, 1.5 and 5 permits are required per pound in the first and second scenarios, respectively.

Ideally, trading ratios would incorporate all available information on how marginal damages from NOx pollution vary across counties, municipalities, or even facilities. I was unable to obtain marginal damage estimates at this level of detail. "Low damage" states are defined to be those that are either completely or marginally in attainment with the federal one hour and eight hour ozone standards (according to the US EPA's "Green Book"). "High damage" states are those that include counties classified as moderate, severe or serious under the one hour and eight hour standards (EPA Green Book). Under exposure-based trading, I assume that a permit is required to offset a pound of NOx in low damage areas; 1.5 permits (or 5 permits in the second scenario) are required in high damage areas.

Defining the baseline

A second set of assumptions have to do with establishing a baseline or benchmark against which to compare simulated emissions under exposure-based trading. Under emissions-based trading, the number of permits distributed equals the total cap on emissions. Assuming perfect compliance, the regulator has complete control over the total amount of pollution that is emitted. Under a trading ratio system, the regulator cannot directly cap emissions. The number of permits distributed equals the permitted damages. The total quantity of permitted emissions will depend on which firms use permits, and which firms invest in pollution reduction. If more permits are used in low

[65] Krupnick et al.(1998) generate trading ratios for a subset of the states affected by the NOx SIP Call. The authors use an urban airshed model to link regional changes in NOx emissions in different regions to regional, population weighted changes in ozone concentrations. They use emissions and meteorological data from three typical five day ozone episodes in 1990 to estimate trading ratios. The authors note that 1990 was a "good" ozone year; their estimates of typical changes in ozone concentrations attributable to sources are conservative. Averaged across typical episodes, ratios range from 1 in low damage areas to 1.5 in high damage areas.

[66] Mauzerall et al (2005) use a comprehensive air quality model (CAMx) to quantify the variable impacts that a fixed quantity of NOx emitted from individual point sources can have on downwind ozone concentrations and resulting population weighted health damages. Simulations were carried out using data from a 10 day period in 1995 (July 7-17). Considering fatality effects only (i.e. ignoring morbidity) and using "off the shelf" estimates of the value of a statistical life, the estimated damage per ton of NOx emissions ranges from 1995 $10,700 to $52,800 depending on ambient temperature and location. This suggests that the appropriate trading ratios in high damage areas could be as large as 5:1. Ratios that take morbidity and environmental damages into account would be even larger.

(high) damage areas, the total amount of pollution will be greater (smaller) for a given cap.

To facilitate a comparison between emissions-based and exposure-based permit market designs, I assume that the cap is defined in terms of emissions in both cases. Put differently, I simulate compliance choices and emissions under exposure-based and emissions-based permit markets that are designed to deliver the same total quantity of seasonal emissions (in terms of pounds of NOx). The emissions predicted by the model conditional on the predicted compliance choices are used as the basis for comparing alternative exposure-based trading outcomes. The emissions-based benchmark outcomes are simulated in the same way that emissions under counterfactual, exposure-based trading are simulated. Appendix D includes a discussion of how this benchmark outcome compares to emissions observed in the first year of permit trading.

D. Simulation

Two sets of simulations are carried out: one which assumes decisions are made by plant managers, and the other which assumes decisions are made at the firm level. The econometric model is used to predict emissions under emissions-based and exposure-based permit trading as follows:

1. The permit price τ is initially set equal to the price that prevailed during the years in which firms were making their compliance decision ($2.25/lb).
2. A vector of coefficients b^r is drawn from the distribution of the random coefficients in the population; r denotes the repetition ($r = 1...1000$).
3. For each unit, expected choice probabilities as defined in [7] are approximated for all compliance available choices. These are conditional on the price τ, b^r, the character and outcomes of observed choices of the corresponding manager (or firm), and the assumed trading ratio R_m.
4. Unit level compliance choices for all choice situations faced by each manager (firm) are predicted. Each unit is assumed to choose the compliance strategy with the highest estimated probability.
5. Seasonal emissions (measured in lbs of NOx) corresponding to the predicted choices are calculated and summed across units.

6. If the total quantity of emissions equals the assumed cap, τ is the equilibrium price and the simulation stops. Equilibrium emissions in high damage areas and low damage areas are calculated.

7. If the total quantity of emissions exceeds (is less than) the cap, τ is increased (decreased) by \$0.01. Steps 3-6 are repeated.[67]

This procedure is repeated 1000 times under the baseline case (emissions-based trading), the conservative exposure-based trading case where $R = 1.5$ in high damage areas, and the less conservative exposure-based trading case where $R = 5$ in high damage areas. Distributions of predicted equilibrium emissions are generated for each scenario. Summary statistics are reported in tables 8 and 9.

If we assume that compliance decisions are made at the facility level (i.e. cost coefficients are allowed to vary across facilities owned by the same parent company) the model predicts an average reduction of 129 tons per day (6 percent) in emissions in the high damage states under the first case ($R = 1.5$), and an average reduction of 457 tons per day (22 percent) in high damage states under the second case ($R = 5$). If we assume that parent companies make compliance decisions, simulation exercises predict reductions of similar magnitude (7 percent and 23 percent, respectively).

These results suggest that the health damages that have resulted (and that will continue for the foreseeable future) from the decision to adopt an emissions-based permit design are non-negligible. Allowing for the fact that the model does over-predict actual emissions (See Appendix D), a 6 to 23 percent decrease in observed emissions in high damage areas translates to moving 92-360 tons of NOx emissions *per day* out of high damage areas into low damage areas, depending on the chosen trading ratios.

VII. Conclusion

This paper provides evidence that generators in restructured electricity markets were less likely to install capital intensive pollution control technology as compared to very similar plants in

[67] If this iterative procedure arrives at a point where adding or subtracting a cent delivers aggregate emissions on either side of the cap, the price that delivers the quantity of emissions just below the cap is chosen to be the equilibrium price. Equilibrium emissions are calculated and the simulation stops.

regulated electricity markets. This result is robust to a variety of specifications.

The relationship between economic regulation in the electricity market and pollution control technology adoption decisions affects permit market efficiency in two ways. First, because the plants with the lowest pollution control costs are not always the ones installing pollution controls, the permit market may fail to minimize the total economic cost of meeting the exogenously determined emissions cap. Whereas a deterministic model that assumes cost minimization and assumes away interstate variation in electricity market regulation predicts that investment in pollution control equipment will be approximately equal in restructured and regulated markets, estimated costs conditional on observed choices suggest that over 60% of investment occurred in regulated markets.

Second, because air quality problems are more severe in states that have restructured their electricity markets, inefficiencies associated with emissions-based trading of a non-uniformly mixed pollutant are exacerbated. In theory, exposure-based permit trading could reduce the efficiency costs of the negative capital bias in restructured electricity markets. The econometric model is used to predict how technology adoption, and thus emissions, would have been different under an exposure-based trading program designed to meet the same total emissions cap. The model predicts that 6-27 percent of permitted emissions (or 92-413 tons of NOx per day, based on observed emissions in 2004) would have been moved out of high damage areas and into low damage areas under a generally defined exposure-based program, relative to an emissions-based program. Recent epidemiological research suggests that a spatial shift in emissions of this magnitude could reduce premature deaths from ozone exposure by hundreds each year. There would also be additional benefits, including reduced morbidity and reduced environmental damages. While this analysis is somewhat limited in how accurately it can measure the precise number of tons of NOx that would move out of high damage areas and into low damage areas under exposure-based trading, the inefficiency of emissions-based permit trading is clear.

The Mercury Rule and the Clean Air Interstate Rule, both finalized in 2005, are scheduled to take effect in 2010. Both will affect electricity generators in both restructured and regulated electricity markets. Both propose to use an emissions-based permit trading program to regulate non-uniformly mixed pollutants. The findings presented here caution against designing permit markets that fail to reflect spatial variation in marginal damages from pollution, particularly when variation in economic regulation across electricity markets is already reducing the probability that pollution controls will be installed in the areas where they deliver the greatest social benefits.

Appendix A: A Model of Compliance Cost Minimization

For all units in the sample, $K_n'(v) < 0; K_n''(v) \geq 0$. For ease of exposition, the compliance decision is represented as a choice of a point on the continuous, convex cost frontier $K_n(v)$.

The Compliance Decision in Restructured Markets

Three ISOs operate centralized power markets in the region regulated by the NBP.[68] All three operate as uniform price auctions wherein the price is set by the marginal bidder. The manager's compliance choice of v_n can affect the unit's position in the dispatch order (relative to other units supplying the market). If the unit is never the marginal (price setting) unit, an increase in v_n will have no effect on the wholesale electricity price.

Let $\bar{P}_n$ represent the average wholesale electricty price paid to unit m. Let ψ_n represent the fraction of variable compliance costs that is not translated into increases in $\bar{P}_n$:

$$1 - \frac{\partial \bar{P}_n}{\partial v_n} = \psi_n. \tag{A1}$$

The compliance choices of plants in this sample will rarely affect the average electricity price $\bar{P}_n$ that the firm receives in the wholesale market because coal-fired generating units are typically infra-marginal. For a unit that is never marginal, $\psi_n = 1$.

The levelized annual compliance cost that the manager of the *nth* plant expects to incur if she chooses choose compliance strategy j is:

$$\begin{aligned} LAC_{nj} &= \psi_n v_{nj} Q_n + l_n K_{nj}, \\ l_n &= \frac{r_n(1+r_n)^{T_n}}{(1+r_n)^{T_n} - 1} \end{aligned}$$

The installation cost K_{ni} is multiplied by the levelized annual cost factor l_n. This yields the annual capital ammortiziation over a period T_{ni} . The annuity interest rate r_n is a weighted average of the cost of debt and the opportunity cost of equity (i.e. the firm's cost of capital).

I assume that the manager chooses v_{ni} to minimize levelized annual compliance costs subject

[68] These are the New York ISO, the New England ISO and the "PJM" (Pennsylvania Jersey Maryland) ISO.

to the constraint that the chosen compliance strategy must lie on the least-cost compliance frontier $K_n(v_{ni})$:

(A2) $$\min_{v} \; LAC_n = \psi_n v q_n + l_n K_n(v),$$

Minimization of the above constrained optimization problem implies:

(A3) $$K_n'(v) = -\frac{\psi_n Q_n}{l_n}$$

The manager will want to choose the point on the compliance cost frontier such that the (negative) slope is equal to the ratio of the cost of an incremental change in variable compliance costs and the cost of an incremental change in fixed compliance costs. [69]

The ratio of the capital cost and variable cost can be interpreted as approximately equal to the firm's discount rate r_n scaled by $\psi_n$when the firm's investment is infintely long:

$$\begin{aligned} LAC_n &= \psi_n v_{nj} + l_n K_{nj}, \\ \frac{dK_n}{dv_{nj}} &= \psi_n \frac{(1+r_n)^{T_n} - 1}{r_n(1+r_n)^{T_n}} \\ \lim_{T_n \to \infty} \frac{dK_{nj}}{dV} &= \psi_n r_n. \end{aligned}$$

For a plant that is always inframarginal and that has an infinitely long investment horizon, the ratio of the variable cost and capital cost coefficient is equal to the firm's discount rate r_n.

Compliance Choices in Unrestructured Markets

I assume that managers at regulated utilities comply with environmental regulations while minimizing compliance costs borne by shareholders (or taxpayers in the case of government owned facilities). Following the example of Fullerton et al.(1997), I define parameters that describe how

[69] This implies that an increase in the cost of capital will, ceteris paribus, be associated with a less capital intensive compliance choice. Similarly, a decrease in ψ_n would lead to a less capital intensive compliance choice.This assumes that restructured markets are closely monitored, so that sellers need to justify bids with operating costs.

compliance costs are shared between ratepayers and shareholders.[70] Let θ_n^V represent the portion of variable compliance costs born by the utility and its shareholders versus the ratepayers. Similarly, let θ_n^K be the portion of capital investments in NOx control technology that the utility cannot pass through to ratepayers.

I assume that the manager chooses v_{ni} to minimize levelized annual compliance costs subject to the constraint that the chosen compliance strategy must lie on the least-cost compliance frontier $K_n(v_{ni})$:

$$\text{(A4)} \qquad \min_v LAC_n = \theta_n^v v Q_n + \theta_n^K l_n K_n(v).$$

Minimization of the above constrained optimization problem implies :

$$\text{(A5)} \qquad K_n'(v) = -\frac{\theta_n^v Q_n}{\theta_n^K l_n}$$

The ratio of the capital cost and variable cost can be interpreted as approximately equal to r_n scaled by the ratio of the cost recovery parameters:

$$\begin{aligned} LAC^{REG} &= \theta^V V + \theta^K l K, \\ \frac{dK}{dV} &= \frac{\theta_n^V}{\theta_n^K l} = \frac{\theta_n^V}{\theta_n^K} \frac{(1+r_n)^{T_n} - 1}{r_n(1+r_n)^{T_n}} \\ \lim_{T\to\infty} \frac{dK}{dV} &= \frac{\theta_n^V}{\theta_n^K} \cdot r_n \end{aligned}$$

If variable and capital costs are treated symmetrically by regulators, this will be r_n. Otherwise, when cost recovery rules favor capital intensive compliance options, the ratio of these model coefficients will overestimate r_n.

Consider two units that face the same compliance cost frontier $K(v)$ and operate at the same production levels but operate in different electricity market environments. Let U denote the firm operating in an unrestructured electricity market and R denote the firm operating in a restructured

[70] There is some evidence that the fixed and variable components of compliance cost have been treated asymmetrically by regulators, so I define different cost recovery parameters for different compliance cost components.

electricity market. If firm R chooses to locate on a steeper portion of $K(v)$, it must be that:

$$\frac{\theta_n^v}{\theta_n^K}\frac{1}{l_R} > \psi_n \frac{1}{l_U}.$$

There are at least three reasons why we might expect this inequality will hold:

1. $\frac{\theta_n^v}{\theta_n^K} > \psi_n$: Rates of return authorized by regulators provide stronger investment incentives in regulated markets as compared to restructured markets.

2. $\frac{1}{l_R} > \frac{1}{l_U}$.: Regulated utilities have higher credit ratings and lower costs of capital on average.

3. *Differences in the option value of waiting*: Managers in regulated markets are assured of cost recovery; there is no uncertainty and thus no option value. To the extent that managers in restructured markets account for real option value when evaluating option alternatives, [A3] will overestimate the slope at the optimal point..

Appendix B: Data Description

Data needed to identify coal units regulated by the NBP

1. U.S. EPA's Clean Air Markets: Program provides a comprehensive list of all the units affected by the NBP (includes the facility name, facility and unit identification numbers, location and contact information).
2. U.S. EPA National Electric Energy System (NEEDS).

Unit-level compliance strategy choices

1. EPA Electronic Data Reporting for the Acid Rain Program/subpart H.
2. Energy Information Administration (EIA).
3. Institute for Clean Air Companies.
4. MJ Bradley & Associates.

Data required to estimate control costs at the unit level

1. U.S. EPA National Electric Energy System (NEEDS).
2. EPA Electronic Data Reporting for the Acid Rain Program/subpart H.
3. U.S. EPA Emissions and Generation Integrated Database (EGRID).
4. Energy Information Administration (EIA) Form 767.
5. Energy Information Administration (EIA) Form 860
6. Platts BaseCase:
7. Raftelis Financial Consultants Water and Wastewater Rate Survey.
8. Bureau of Labor Statistics: Regional estimates of boilermaker and construction wages.
9. Personal Correspondence: Representatives from the major coal-fired boiler manufacturers (Alstom Engineering, Babcock Power, Foster Wheeler, Riley Power Inc.) provided valuable information about the technical specifications of the boilers in the sample De-NOx Technologies LLC provided data on reagent and reagent transportation costs. Other technical assistance was provided by Cichanowicz Consulting Engineers LLP.

Permit Price/Transaction Data

1. Evolution Markets LLC

Estimates of anticipated post-retrofit NOx emissions rates (conditional on boiler characteristics) constructed using the following sources:

1. Biewald, B., J. Cavicchi, T. Woolf and D. Allen. 2000. "Use of Selective Catalytic Reduction for control of NOx Emissions from Power Plants in the U.S." *Synapse Energy Economics Inc.*

2. Cichanowicz, J.E. 2004. "Why are SCR costs still rising?". *Air Quality Control*, 148(3): 32.

3. Electric Policy Research Institute. 1999. "Application of Methodology for Identification of Least Cost NOx Control Combinations."

4. Electric Policy Research Institute. 1999. UMBRELLA: "Software for Assessing NOx Control Technology Combinations, Version 1.0."

5. Farzan, H. G.J. Maringo, D.W. Johnson, D.K. Wong . 1999."B&W's Advances on Cyclone NOx Control via Fuel and Air Staging Technologies", EPRI_DOE_EPA Combined Utility Air Pollutant Control Symposium, Atlanta GA.

6. Staudt, J. "Technologies and Cost Effectiveness of Post-Combustion NOx Controls." Andover Technology Partners, http://www.netl.doe.gov/publications/proceedings/99/99scr-sncr/staudt.pdf.

7. Steitz, T.H., R.W. Cole. 1999. "Field Experience in Over 30,000 MW of Wall Fired Low NOx Installations." Foster Wheeler Energy Corporation.

8. US Department of Energy. 2002."Full-Scale Demonstration of Low-NOx Cell Burner Retrofit." http://www.netl.doe.gov/cctc/summaries/clbrn/cellburnerdemo.html.

9. US Environmental Protection Agency. 2002. Documentation Supplement for EPA Modeling Applications (V.2.1) Using the Integrated Planning Model. Office of Air and Radiation. Washington D.C.

10. US Environmental Protection Agency. 1999. Regulatory Impact Analysis for the Final Section 126 Petition Rule, Office of Air and Radiation, Washington DC.

11. US Environmental Protection Agency. 1998. Regulatory Impact Analysis for the NBP, Office of Air and Radiation, Washington DC.

12. US Environmental Protection Agency. 1998, Feasibility of Installing NOx Control Technologies by May 2003, Office of Atmospheric Programs, Acid Rain Division, Research Triangle Park, NC.

13. US Environmental Protection Agency. 1998. Regulatory Impact Analysis for the NBP, FIP and Section 128 Petitions, Office of Air Quality Planning and Standards, Office of Atmospheric Programs, Washington D.C.

14. US Environmental Protection Agency. 1998. Analyzing Electric Power Generation under the CAAA. Office of Air and Radiation. Washington D.C.

Appendix C: Deriving the Conditional Logit Choice Probabilities Implied by Cost Minimization

It is straightforward to show that for additive, iid extreme value (Type I) errors, the assumption of cost minimization does not yield the standard CL choice probabilities due to the asymmetry of the assumed distribution. In the standard Random Utility Maximization (RUM) logit model, the assumption of an additive extreme value error term is motivated by a desire for simple closed-form expressions for choice probabilities. Here I show that, in the context of cost minimization, assuming that the extreme value term is subtracted from (versus added to) the deterministic component implies equally convenient expression for choice probabilities. This closely follows the derivation of the standard RUM choice probabilities in Train(2003).

The unit (denoted n) chooses from among J_n compliance alternatives. The cost that the unit associates with each alternative is comprised of a deterministic component and a stochastic component:

$$C_{ni} = \beta_m X_{ni} - \varepsilon_{ni},$$

where ε_{ni} is assumed to be independently, identically distributed type I extreme value. To derive the choice probabilities, I assume that the unit chooses the compliance option that minimizes anticipated compliance costs.(For ease of notation, the n subscript on the coefficient vector β is dropped). Let P_{ni} be the probability that unit n chooses alternative i :

$$\begin{aligned} P_{ni} &= \text{Prob } (\beta' X_{ni} - \varepsilon_{ni} < \beta' X_{nj} - \varepsilon_{nj} \quad \forall\, j \neq i) \\ &= \text{Prob } (\varepsilon_{nj} < \beta' X_{nj} - \beta' X_{ni} + \varepsilon_{ni} \quad \forall\, j \neq i) \end{aligned}$$

The expression for the conditional choice probability :

$$\begin{aligned} P_{ni}|\varepsilon_{ni} &= \prod_{j \neq i} F(\beta' X_{nj} - \beta' X_{ni} + \varepsilon_{ni}) \\ &= \prod_{j \neq i} \exp(-\exp(-(\beta' X_{nj} - \beta' X_{ni} + \varepsilon_{ni}))) \end{aligned}$$

Unconditional choice probabilities are obtained by integrating over the distribution of ε_n :

$$\begin{aligned} P_{ni} &= \int_{\varepsilon=-\infty}^{\infty} \prod_{j \neq i} \exp(-\exp(-(\beta' X_{nj} - \beta' X_{ni} + \varepsilon_{ni}))f(\varepsilon_n)\, d\varepsilon_n \\ &= \int_{s=-\infty}^{\infty} \prod_{j \neq i} \exp(-\exp(-(\beta' X_{nj} - \beta' X_{ni} + s)) \exp(-s) \exp(-\exp(-s))\, ds \end{aligned}$$

Note that $\exp(-\exp(-(\beta' X_{nj} - \beta' X_{nj} + s))) = \exp(-\exp(-s))$. Making this substitution:

$$
\begin{aligned}
P_{ni} &= \int\limits_{s=-\infty}^{\infty} \prod_j \exp(-\exp(-(\beta' X_{nj} - \beta' X_{ni} + s))\exp(-s)ds \\
&= \int\limits_{s=-\infty}^{\infty} \exp - \sum_j \exp(-(\beta' X_{nj} - \beta' X_{ni} + s))\exp(-s)ds \\
&= \int\limits_{s=-\infty}^{\infty} \exp(-\exp(-s)) \sum_j \exp(-(\beta' X_{nj} - \beta' X_{ni}))\exp(-s)ds
\end{aligned}
$$

We define a variable t such that $t = \exp(-s) \Rightarrow dt = -\exp(-s)ds$. Making this substitution:

$$
P_{ni} = \int\limits_{s=0}^{\infty} \exp(-t \sum_j \exp(-(\beta' X_{nj} - \beta' X_{ni}))dt
$$

Evaluating this integral, we are left with:

$$
P_{ni} = \frac{1}{\sum\limits_j \frac{\exp(\beta' X_{ni})}{\exp(\beta' X_{nj})}}
$$

An alternative way of expressing this conditional choice probability:

$$
P_{ni} = \frac{\frac{1}{\exp(\beta' X_{ni})}}{\sum\limits_j (\frac{1}{\exp(\beta' X_{nj})})} = \frac{\exp(-\beta' X_{ni})}{\sum\limits_j \exp(-\beta' X_{nj})}
$$

Appendix D: Comparing predicted and observed emissions

Significant discrepancies exist between observed emissions during the first ozone season and emissions predicted by the model under emissions-based permit trading. Table A1 compares observed emissions from the first ozone season of the NBP (2004) to the emissions predicted by the model (I use manager-specific cost coefficients here).

Table A1: Observed and Predicted Average NOx Emissions (tons per day) by Market Type

	Observed (2004 season)	Predicted | Observed Choices	Predicted | Predicted Choices
			(BASELINE)
Restructured markets NOx emissions (tons/day)	1662	2272	2349 (64)
Regulated markets NOx emissions (tons/day)	1592	2022	1999 (64)
Total NOx emissions (tons/day)	3254	4294	4348 (6)
% Emissions in restructured markets	51%*	53%	54% (0.5%)

Notes: Standard deviations are in parentheses.

The second and third columns report predicted emissions conditional on observed choices and conditioned on simulated choices, respectively. Although the model is quite accurate in predicting compliance choices, it does a poor job of predicting emissions. Predicted emissions (based on predicted compliance choices) are 34% higher than observed emissions overall and over 40% higher in states with restructured electricity markets.

A closer look at the data reveals three reasons for these discrepancies. First, the model assumes that emissions rates (measured in lbs NOx/mmbtu) for those units that choose not install any pollution controls will equal the unit's average, historic ozone season emissions rate (i.e. 1999-2002). In fact, emissions rates at units that chose to rely entirely on the permit market for compliance fall by an average of 21% in the first ozone season, relative to past summers. This relationship does not differ significantly across electricity market types.[71] Emissions rates at these plants were likely reduced by changing boiler conditions so as to reduce NOx formation during combustion.

Second, the unit-specific, technology-specific, post-retrofit NOx removal rates assumed by the model also appear to have been conservative. These are the same estimates that were made available to plant managers while they were making their compliance decisions. Among units that adopted some pollution control technology other than SCR, observed post-retrofit NOx emissions rates are, on average, 27% below predicted post-retrofit NOx rates. Among units adopting SCR,

[71] The average decrease in NOx rates is 22% (with a standard deviation of 26%) in regulated markets and 19% in restructured markets (with a standard deviation of 21%).

observed post-retrofit emissions rates are, on average, 41% below predicted rates in restructured electricity markets and 28% below predicted rates in regulated markets. The reason for the difference across electricity market types is that several plants installing SCR reportedly were unable to complete their SCR retrofits in time for the first ozone season; most of these are in regulated electricity markets. Consequently, observed NOx rates in the summer of 2004 greatly exceeded the predicted NOx rates at these plants. The emissions rates at these plants, and the proportion of permitted NOx emissions in states with regulated electricity markets, should decline in future ozone seasons as SCR retrofits are completed.

Finally, assumptions about unit-level heat rates (measured in mmbtu/kWh) also underestimate ex post observed unit-level performance. The model assumes that future unit-level heat rates will equal those observed in previous summers. On average, units performed more efficiently in the summer of 2004 than in past ozone seasons. When observed heat rates are regressed on predicted heat rates and NOx control technology dummies, the coefficient on predicted heat rates is 0.91 with a standard error of 0.01. None of the technology dummies are statistically significant. Results do not change when regression equations are estimated separately for regulated and restructured markets.

Because observed emissions are significantly lower than the emissions predicted by the model, comparing emissions predicted under counterfactual exposure-based policy simulations with observed emissions would be uninformative and misleading. Instead, baseline emissions (i.e., the emissions associated with the observed, emissions-based permit trading program) are simulated in the same way that emissions under counterfactual, exposure-based trading are simulated.

References

[1] Allenby, Greg and Peter Rossi. 1990. "Marketing Models of Consumer Heterogeneity." *Journal of Econometrics*, 89: 57-78.

[2] Allison, Paul.D. 1999. "Comparing Logit and Probit Coefficients Across Groups." *SMR/Sociological Methods and Research*, 28: 186-208.

[3] Ando, Amy and Karen Palmer. 1998. "Getting on the Map: The Political Economy of State-Level Electricity Restructuring." *Resources for the Future Discussion Paper 98-19-REV.*

[4] Arimura, Toshi. 2002. "An Empirical Study of the SO_2 Allowance Market: Effects of PUC Regulations." *Journal of Environmental Economics and Management,* 44: 271-289.

[5] Averch, Harvey and Leland Johnson. 1962. "Behavior of the Firm under Regulatory Constraint." *American Economic Review,* 52 (5): 1053-1069.

[6] Bailey, Elizabeth M. 1996. "Allowance Trading Activity and State Regulatory Rulings: Evidence from the U.S. Acid Rain Program." MIT *Center for Energy and Environmental Policy Research WP-96002*,

[7] Bailey, Elizabeth M. and John C. Malone. 1970. "Resource Allocation and the Regulated Firm." *Bell Journal of Economics and Management Science*, Spring (1): 129-142.

[8] Baumol, William J. and Wallace E. Oates. 1971. "The Use of Standards and Prices for Protection of the Environment." *Swedish Journal of Economics*, 73: 42-54.

[9] Bhat, Chandra. 2003. "Econometric Choice Formulations: Alternative Model Structures, Estimation Techniques, and Emerging Direction." Econometric Models of Choice: Formulation and Estimation, IATBR Conference, Lucerne, Switzerland.

[10] Bhat, Chandra. 1998. "Accommodating variations in responsiveness to level-of-service variables in travel mode choice models", *Transportation Research A*, 32: 455–507.

[11] Bohi, Douglas R. and Dallas Burtraw. 1992."Utility Investment Behavior and the Emission Trading Market", *Resources and Energy,* 14: 129-153.

[12] Boyes, William. "An Empirical Examination of the Averch-Johnson Effect. 1976." *Economic Inquiry*, 14(1): 25-35.

[13] Brownstone, David and Kenneth E. Train. "Forecasting new product penetration with flexible substitution patterns." *Journal of Econometrics,* 89: 109–129.

[14] Bushnell, Jim and Catherine Wolfram. 2005. "Ownership Change, Incentives and Plant Efficiency: The Divestiture of U.S. Electric Generation Plants." *UC Energy Institute, CSEM Working Paper 140.*

[15] *Business Wire.* "Fitch Expects to Rate PSI Energy First Mortgage Bonds 'A-.'"June 18, 2001.

[16] *Business Wire.* "Fitch Revises Northeast Utilities' Rating Outlook to Negative." May 19, 2003.

[17] *Business Wire.* " Fitch Ratings Affirms Cinergy & Subsidiaries; Outlook Stable." April 13, 2004a.

[18] *Business Wire.* "Fitch Upgrades Alabama Power Company Sr Unsecured Rating to 'A+'." Nov 12, 2004b.

[19] *Charleston Gazette*. "American Electric Power to Invest in Pollution Control at West Virginia Plants." Feb. 28, 2004.

[20] Coase, Ronald H. 1960. "The Problem of Social Cost." *Journal of Law and Economics.* 3: 1-44.

[21] Crocker, Thomas D. 1966 "The Structuring of Atmospheric Pollution Control Systems." *The Economics of Air Pollution*, ed. H. Wolozin, 61-86. New York, W. W.Norton & Co.

[22] Cichanowicz, John E. 2004. "Why are SCR costs still rising?". *Air Quality Control,* 148(3): 32.

[23] Coggins, John and Vincent H. Smith. 1993. "Some Welfare Effects of Emissions Allowance Trading in a Twice-Regulated Industry." J*ournal of Environmental Economics and Management*, 25: 274-297.

[24] Courville, Leon. 1974. Regulation and Efficiency in the Electric Utility Industry, *Bell Journal of Economics*, 5(1): 53-74.

[25] Dales, John H. 1968. *Pollution, Property and Prices.* Toronto, University of Toronto Press.

[26] Duggan, John and Joanne Roberts. 2002. "Implementing the Efficient Allocation of Pollution." *American Economic Review*, 92(4): 1070-1078.

[27] Electric Policy Research Institute. 1999a. *Application of Methodology for Identification of Least Cost NOx Control Combinations.*

[28] Electric Policy Research Institute. 1999b. *UMBRELLA: Software for Assessing NOx Control Technology Combinations*, Version 1.0.

[29] *Electricity Daily.* "Alabama Power finishes second phase of SCR program." June 5, 2003.

[30] Energy Information Administration (EIA). 1999. "Status of Electricity Industry Restructuring." <http://www.eia.doe.gov/cneaf/ electricity/chg_str/restructure.pdf>.

[31] Fabrizio, Kira, Nancy Rose, Catherine Wolfram. 2006. "Does Competition Reduce Costs? Assessing the Impact of Regulatory Restructuring on U.S. Electric Generation Efficiency." *CSEM Working Paper 135,* UC Energy Institute.

[32] Farrell, Alex., Roger Raufer, and Robert Carter. "The NOX Budget: Market-Based Control of Tropospheric Ozone in the Northeastern United States." *Resource & Energy Economics,*21: 103-124..

[33] Fullerton, Don, Shaun McDermott and Jonathan P. Caulkins. 1997. "Sulfur Dioxide Compliance of a Regulated Utility." Journal of Environmental Economics and Management, 34(1): 32-53.

[34] Greene,William. 2003. "Interpreting Estimated Parameters and Measuring Individual Heterogeneity in Random Coefficient Models." Department of Economics, Stern School of Business, New York University.

[35] Hahn, Robert W.1990. "Regulatory Constraints on Environmental Markets." *Journal of Public Economics,* 42: 149-75.

[36] Hayashi, Paul M. and Trapani, John M. 1976. "Rate-of-Return Regulation and the Regulated Firm's Choice of Capital-Labor Ratio: Further Empirical Evidence on the Averch-Johnson Model." *Southern Economic Journal*, 42: 384-398.

[37] Hensher, David A. and William Greene. 2003. "The Mixed Logit model: The State of Practice." *Transportation*, 30 (2):133-176.

[38] Himes, Richard. Electric Power Research Institute. Telephone conversation, July 12, 2004.

[39] Hoetker, Glen. 2003. "Confounded Coefficients: Accurately Comparing Logit and Probit Coefficients Across Groups." *Working paper, College of Business, Universioty of Illinois at Urbana-Champaign.*

[40] Joskow, Paul, Robert Schmalensee, et al. 1998. The Market for Sulfur Dioxide Emissions, *The American Economic Review,*88(4): 669-685.

[41] Joskow. Paul. 1974. "Inflation and Environmental Concern: Structural Change in the Process of Public Utility Price Regulation," *Journal of Law and Economics*, 17(2): 291-327.

[42] Keohane, Nathaniel. 2005. Environmental Policy and the Choice of Abatement Technique: Evidence from Coal-Fired Power Plants, *mimeo, Yale University*.

[43] Krupnick, Alan., Virginia McConnell, Matthew Cannon, Terrell Stoessell and Mark Batz. "Cost Effective NOx Control in the Eastern United States." *Resources for the Future Discussion Paper*, April 2000.

[44] Krupnick Alan, Wallace Oates, EricVan de Verg . 1983. On marketable air-pollution permits: the case for a system of pollution offsets. J*ournal of Environmental Economics and Management*, 10(3): 233-47.

[45] Layton, David and G. Brown. 2000. "Heterogeneous Preferences Regarding Global Climate Change." *Review of Economics and Statistics*, 82: 616-24.

[46] Leahy, Kevin. 2005. Personal correspondence.

[47] Lin, Cynthia Y.C., Daniel Jacob, and Arlene Fiore. 2001. "Trends in exceedances of the ozone air quality standard in the continental United States, 1980-1998." *Atmospheric Environment*, 35: 3217-3228.

[48] Mansur, Erin T. 2004. "Environmental regulation in oligopoly markets: A study of electricity restructuring." *POWER Working Paper,* UC Energy Institute,.

[49] Mauzerall, Denise L., Babar Sultan, Namsong Kim and David Bradford. 2005. "NOx emissions from large point sources: variability in ozone production, resulting health damages and economics costs." *Atmospheric Environment*, 39 (16): 2851-2866.

[50] McFadden, Daniel. 1973. "Conditional Logit Analysis of Qualitative Choice Behavior." *Frontiers in Econometrics.* P. Zarembka ed. Academic Press.

[51] McFadden, Daniel and Kenneth E. Train. 2000. "Mixed MNL Models for Discrete Response." J*ournal of Applied Econometrics,* 15: 447-470.

[52] "North Carolina utility to spend $175 M to cut pollution from plants." *Megawatt Daily*, June 10, 2003.

[53] "Top Projects Completed." *Midwest Construction,* 8(6) p 15. June 1, 2005.

[54] Montgomery, W. David. 1972. Markets in Licenses and Efficient Pollution Control Programs, *Journal of Economic Theory,*5(3): 395-418.

[55] Musatti, Daniel. US Environmental Protection Agency. Email correpondence. June 7, 2004.

[56] National Association of Regulatory Utility Commissioners (NARUC).*A Survey of State Incentives Encouraging Improved Environmental Performance of Base-Load Electric Generation Facilities: Policy and Regulatory Initiatives*, 2004.

[57] "Extra Credits Will Keep Fuel Switching Down." *Natural Gas Week.* April 23, 2004.

[58] Ozone Transport Assessment Group (OTAG). *Telling the OTAG Ozone Story with Data.* OTAG Air Quality Analysis Workgroup, June 1997.

[59] "Hoosier Energy to spend up to $80 million to install SCR." *Platts Utility Environment Report,* July 16, 1999, 2.

[60] "LG&E units seek regulatory approval to add SCR in coal plants for $300M." *Platts Utility Environment Report,* April 21, 2000a.

[61] "Sigeco approved to pass cost of SCR at Culley and Brown to Ratepayers." *Platts Utility Environment Report,* September 7, 2000b.

[62] "AEP to spend $107 Million for SCR system at Big Sandy plant in KY." *Platts Utility Environment Report*, April 20, 2001a.

[63] "TVA charts 10 year $2.5 billion plan to cut emissions in Alabama, Kentucky and Tennessee." *Platts Utility Environment Report*, October 19, 2001b.

[64] "Cost recovery riders for Vectren's NOx upgrades take effect in Indiana." *Platts Utility Environment Report,* February 22, 2002a.

[65] "AEP opts for SCR at 800 MW station under its $1.6 billion NOx control plan." *Platts Utility Environment Report,* June 28, 2002b, pp. 9.

[66] "PSI gets nod from Indiana regulators for $500 Million NOx reduction plan." *Platts Utility Environment Report,* July 26, 2002c, pp. 9.

[67] "Indiana regulators approve Wabash Valley plan to finance upgrades at coal plant." *Platts Utility Environment Report ,* August 23, 2002d, pp. 7.

[68] "Citing Credit Woes, Merchant Needs, Midwest shelves Powerton SCR Project. " *Platts Utility Environment Report ,* Nov. 29, 2002e.

[69] "IP&L granted 8% rate of return on $260M clean coal projects." *Platts Utility Environment Report ,* November 29, 2002f, pp. 4.

[70] "PSI gets nod from Indiana regulators for $500 Million NOx reduction plan." *PR Newswire*, February 8, 2002.

[71] Revelt, David and Kenneth E. Train. 1998. "Mixed Logit with Repeated Choices." *Review of Economics and Statistics,* 80: 647–657.

[72] Revelt, David and Kenneth E. Train. 2000. "Specific Taste Parameters and Mixed Logit." *Working Paper No. E00-274,*, Department of Economics, University of California, Berkeley.

[73] Rose, Nancy and Paul Joskow. 1984. "The Diffusion of New Technologies: Evidence from the Electric Utility Industry," *RAND Journal of Economics*, 21:354-373.

[74] Rossi, Peter, Robert McCulloch and Greg Allenby. 1996. "On the Value of Household Purchase History Information in Target Marketing." *Marketing Science*, 15: 321–340.

[75] Schmalensee, Richard, Paul Joskow et al. 1998. "An Interim Evaluation of Sulfur Dioxide Emissions Trading." *The Journal of Economic Perspectives,* 12(3): 53-68.

[76] Shadbegian, Ronald J., Wayne Gray and Cynthia Morgan. 2006. "A Spatial Analysis of the Consequences of the SO_2 Program." Preliminary draft. Cited with permission.

[77] Senate Committee on Energy and Natural Resources. "Financial Conditions of the Energy Market; Statement of the Edison Electric Institute." March 4, 2003.

[78] Sotkiewicz, Paul M. 2000. The Impact of State-Level Public Utility Commission Regulation on the Market for Sulfur Dioxide Allowances, Compliance Costs, and the Distribution of Emissions, Ph.D. Dissertation, Department of Economics, University of Minnesota.

[79] "KY Co-op gets approcal to spend $140 M on SCR pollution control technology." *Southeast Power Report,* November 27, 2000.

[80] Spann, Robert. 1974. "Rate of Return Regulation and Efficiency in Production: An Empirical Test of the Averch-Johnson Thesis." *Bell Journal of Economics*,5:38-52.

[81] Srivastava, Ravi. US Environmental Protection Agency. Telephone interview with author. May 12, 2004.

[82] Stavins, Robert N. 1998. What Can We Learn from the Grand Policy Experiment? Lessons from SO2 Allowance Trading. *The Journal of Economic Perspectives,* 12(3): 69-88.

[83] Tietenberg, Tom H. 1974. "The Design of Property Rights for Air Pollution Control." *Public Policy*, 27(3): 275-292.

[84] Train, Kenneth E. 1999. "Halton Sequences for Mixed Logit." Manuscript, Department of Economics, University of Berkeley.

[85] Train, Kenneth E. , David Revelt and Paul Ruud. 1999. *Mixed Logit Estimation Routine for Panel Data.*

[86] Train, Kenneth E. 2003. *Discrete Choice Methods with Simulation.* 1st ed. Cambridge: Cambridge University Press.

[87] Train, Kenneth E. and Garrett Sonnier. 2005. "Mixed Logit with Bounded Distributions of Correlated Partworths". Chapter 7 in *Applications of simulation methods in environmental and resource economics,* A. Alberini and R. Scarpa (eds.).Springer Publisher, Dordrecht, The Netherlands.

[88] Train, Kenneth E. and Melvyn Weeks. 2005. "Discrete Choice Models in Preference Space and Willingness-to-Pay Space." Chapter 1 in *Applications of Simulation Methods in Environmental Resource Economics*, A. Alberini and R. Scarpa, eds., Springer Publisher: Dordrecht, The Netherlands.

[89] US Department of Energy. 2005. "Full-Scale Demonstration of Low-NOx Cell Burner Retrofit." <http://www.netl.doe.gov/cctc/summaries/clbrn/cellburnerdemo.html>.

[90] US Environmental Protection Agency. 2005. *Evaluating Ozone Control Programs in the Eastern United States: Focus on the NOx Budget Trading Program, 2004.* Office of Air and Radiation, Office of Air Quality Planning and Standards, Washington DC.

[91] US Environmental Protection Agency. *Greenbook.* http://www.epa.gov/air/oaqps/greenbk/o8index.htm

[92] US Environmental Protection Agency. 2003. *Documentation Supplement for EPA Modeling Applications (V.2.1.6) Using the Integrated Planning Model.* Office of Air and Radiation. Washington D.C.

[93] US Environmental Protection Agency. 1999. *Regulatory Impact Analysis for the Final Section 126 Petition Rule*, Office of Air and Radiation, Washington DC.

[94] US Environmental Protection Agency. 1998a. *Regulatory Impact Analysis for the NBP*, Office of Air and Radiation, Washington DC.

[95] US Environmental Protection Agency. 1998b. *Feasibility of Installing NOx Control Technologies by May 2003,* Office of Atmospheric Programs, Acid Rain Division, Research Triangle Park, NC.

[96] US Environmental Protection Agency. 1998c. *Regulatory Impact Analysis for the NBP, FIP and Section 128 Petitions,* Office of Air Quality Planning and Standards, Office of Atmospheric Programs, Washington D.C.

[97] US Environmental Protection Agency. 1998d. *Analyzing Electric Power Generation under the CAAA.* Office of Air and Radiation. Washington D.C.

[98] Van Doren, Peter and Jerry Taylor. 2004. Rethinking Electricity Restructuring, *Cato Policy Analysis* no. 530.

[99] Whiteman, Chad. Institute for Clean Air Companies. Personal Correspondence, October 2005.

[100] World Health Organization. 2003. "Health Aspects of Air Pollution with Particulate Matter, Ozone and Nitrogen Dioxide." *WHO Working Group,* Bonn, Germany 13–15.

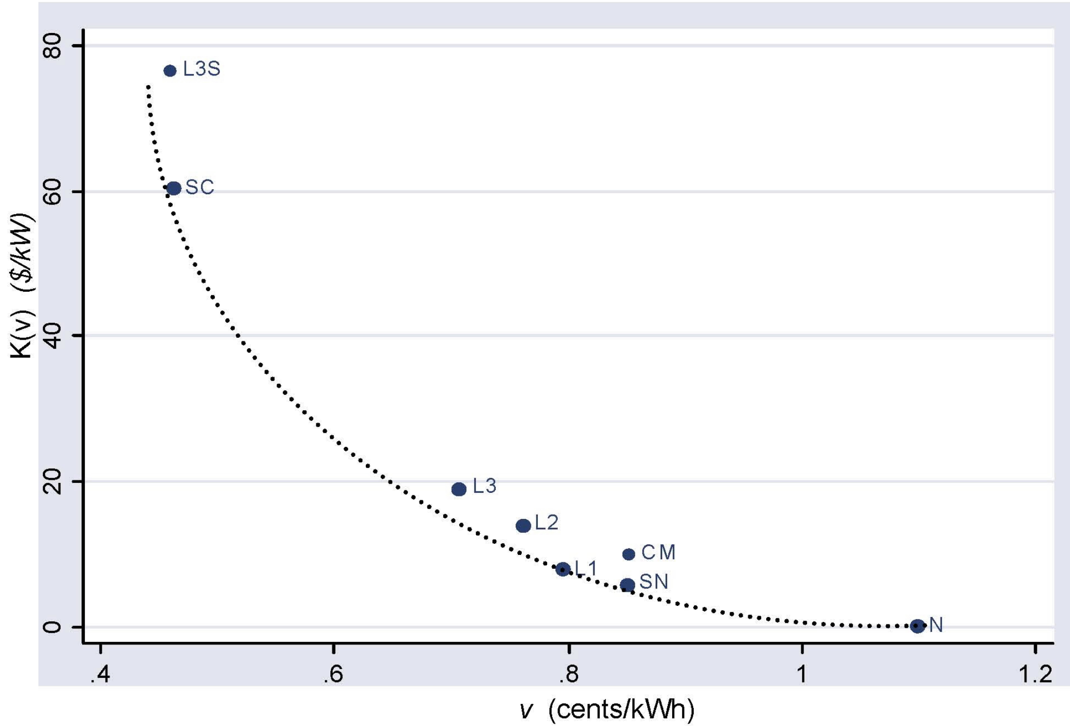


Figure 1: Estimated NOx Control Costs for a 512 MW T-Fired Boiler

Strategy code	Technology	lbs NOx/mmBtu
N	No Retrofit	0.42
SN	Selective Non-Catalytic Reduction (SNCR)	0.34
CM	Combustion Modification	0.33
L1	Low NOx Burners with overfire air option 1	0.31
L2	Low NOx Burners with overfire air option 2	0.28
L3	Low NOx Burners with overfire air options 1&2	0.26
SC	Selective Catalytic Reduction (SCR)	0.13
L3S	L3 + SCR	0.11

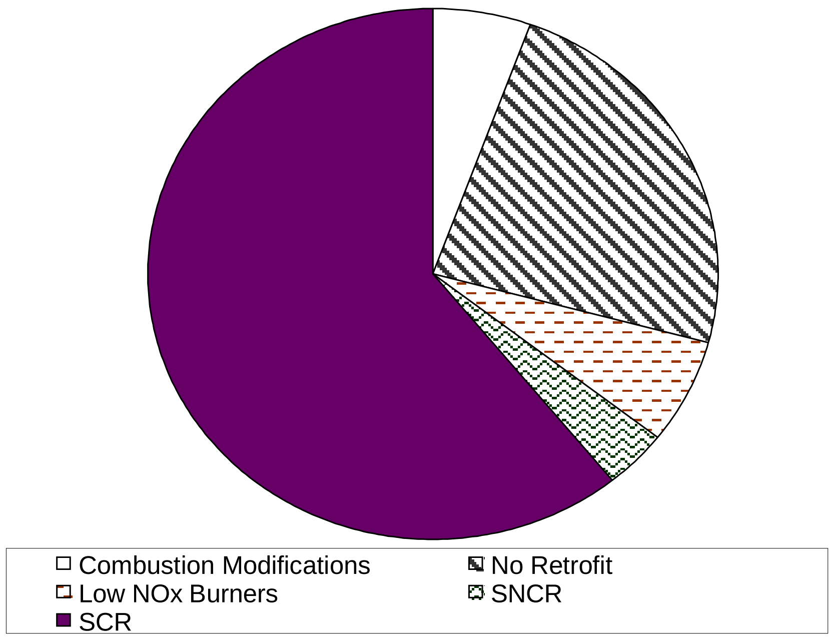


Figure 2a: Compliance Choices of Units in Regulated Markets

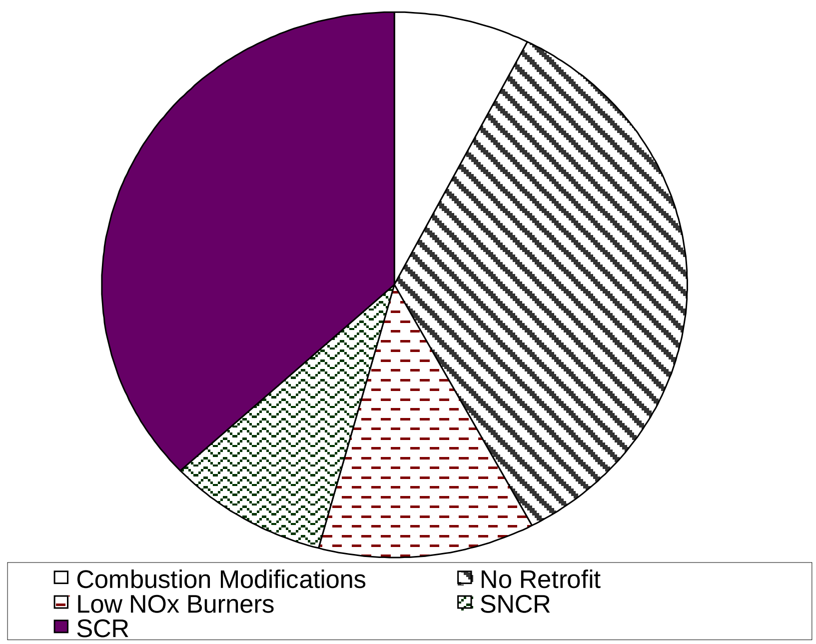


Figure 2b: Compliance Choices of Units in Restructured Markets

Table 1: Summary Statistics by Electricity Market Type

Variable	Restructured	Regulated
# Units	310	322
# Facilities	113	108
Capacity (MW)	275	268
	(243)	(258)
Pre-retrofit NOx emissions (lbs/mmBtu)	0.50	0.54
	(0.21)	(0.22)
Pre-retrofit summer capacity factor (%)	64	67
	(16)	(13)
Pre-retrofit heat rate (kWh/btu)	11,376	11,509
	(2153)	(1685)
Unit Age (years)	37	36
	(11)	(11)

Notes: Standard deviations in parentheses. Summary statistics generated using the data from the 632 units used to estimate the model.

Table 2: Choice Set Summary Statistics by Electricity Market Type

Variable	Restructured	Regulated
# Choices	6.8	6.6
	(1.8)	(1.7)
Combustion Modification	75%	72%
LNB +OFA	36%	32%
SNCR	92%	90%
SCR	100%	100%

Table 3: Compliance Cost Summary Statistics for Commonly Selected Control Technologies

Technology	Capital Cost ($/kW)		Per kWh operating costs (cents/kWh)	
	Restructured	Regulated	Restructured	Regulated
Combustion Modification	12.61	12.21	0.94	1.06
	(4.87)	(4.24)	(0.38)	(0.39)
Low NOx Burners w/ OFA	29.72	31.16	0.64	0.64
	(13.83)	(20.55)	(0.20)	(0.16)
SNCR	16.60	19.16	0.97	1.03
	(14.41)	(21.88)	(0.41)	(0.38)
SCR	70.36	72.90	0.52	0.54
	(21.02)	(25.52)	(0.31)	(0.19)

Notes: Standard deviations are in parentheses.

Table 4. Conditional and Random Parameters Logit Results

	Conditional Logit Model		RCL Model	
	Restructured	Regulated	Restructured	Regulated
	Technology Type Constants			
α_{POST}	-1.89**	-2.63**	-1.35*	-3.39**
	(0.34)	(0.38)	(0.52)	(0.59)
α_{CM}	-1.81**	-2.20**	-1.87**	-2.48**
	(0.26)	(0.28)	(0.30)	(0.32)
α_{LNB}	-1.86**	-2.15**	-1.55**	-2.49**
	(0.33)	(0.29)	(0.37)	(0.31)
	Cost Variables			
Annual compliance costs (V) ($100,000)	-0.30**	-0.31*	-1.21**	-1.00**
	(0.09)	(0.15)	(0.26)	(0.21)
Capital cost (K) ($100,000)	-0.06**	0.02	-0.53**	-0.16
	(0.02)	(0.06)	(0.12)	(0.10)
K*Age	-0.003	-0.002	-0.22**	-0.11*
	(0.002)	(0.003)	(0.06)	(0.05)
Cholesky 1 (σ_V)	–		-1.42**	-0.51**
	–		(0.30)	(0.16)
Cholesky 2 (σ_K)	–		0.30**	0.14**
			(0.08)	(0.05)
Cholesky 3 (off diagonal)	–		0.04	0.04
			(0.11)	(0.07)
# units	310	322	310	322
# facilities	113	108	113	108
Log-likelihood	-431.2	-387.1	-359.4	-326.3
LR Test	compare to technology constants		compare to logit	
	103.94**	211.71**	143.66**	121.64**

Notes: Robust standard errors are in parentheses. *Indicates significance at 5%. **Indicates significance at 1%.

Table 5: Expected Means and Standard Deviations of Manager Specific Coefficient Distributions

Coefficient	Restructured		Regulated	
	Population parameter estimate	Average of conditional parameter estimates	Population parameter estimate	Average of conditional parameter estimates
Annual operating cost (V)	-1.21**	-1.13	-1.00**	-1.00
($100,000)		(1.00)		(0.33)
Capital cost (K)	-0.53**	-0.54	-0.16	-0.16
($100,000)		(0.19)		(0.10)
Elements of the Cholesky factor L of Ω				
Cholesky 1	-1.42**	-0.94	0.51**	0.40
(σ_V)		(0.30)		(0.07)
Cholesky 2	0.30**	0.23	0.14**	0.11
(σ_K)		(0.04)		(0.02)
Cholesky 3	0.04	0.07	0.04	0.002
(off diagonal)		(0.04)		(0.01)
# plants		113		108

Notes: Standard deviations are in parentheses. *Indicates significance at 5%. **Indicates significance at 1%.

Table 6: Average Own Capital Cost and Own Annual Compliance Cost Elasticities for Commonly Selected Technologies

Technology	Own capital cost elasticities		Own annual cost elasticities	
	RESTRUCTURED	REGULATED	RESTRUCTURED	REGULATED
Combustion	-1.03	-0.25	-4.63	-4.40
Modification	(0.81)	(0.33)	(7.37)	(5.02)
Low NOx Burners	-1.25	-0.49	-3.75	-2.18
with overfire air	(1.40)	(0.32)	(4.01)	(1.34)
No retrofit	–	–	-10.02	-8.19
			(18.16)	(13.50)
SCR	-5.74	-1.33	-1.75	-1.34
	(4.02)	(1.15)	(3.23)	(1.64)
SNCR	-1.07	-0.27	-7.56	-6.96
	(0.65)	(0.38)	(14.09)	(8.98)

Notes: These elasticities are calculated using the point estimates of the means of the conditional coefficient distributions. Standard deviations are in parentheses

Table 7: Alternative RPL Specification Results

	Restructured	Regulated
Annual compliance	-0.65**	-0.711**
costs (V)	(0.15)	(0.16)
($100,000)		
Capital cost	-0.21**	-0.06
(K)	(0.08)	(0.05)
($100,000)		
K*Age	-0.05	-0.07*
	(0.03)	(0.03)
Cholesky 1	0.52**	0.27**
(σ_V)	(0.20)	(0.06)
Cholesky 2	0.21**	0.07*
(σ_K)	(0.08)	(0.03)
Cholesky 3	0.10	0.04
(off diagonal)	(0.06)	(0.03)
# units	310	322
# facilities	50	45
Log-likelihood	-395.59	-351.01

Notes: Robust standard errors are in parentheses. *Indicates significance at 5%. **Indicates significance at 1%.

Table 8: Testing the Independence of Ozone Season Production and Compliance Strategy Choice

	Restructured	Regulated
Past ozone season	1.00**	1.03**
production	(0.04)	(0.01)
(average kWh)		
Past production x	-0.12	-0.04
Combustion modification	(0.07)	(0.04)
Past production x	0.04	-0.04
low NOx burners	(0.07)	(0.05)
Past production x	0.09*	-0.00
SCR	(0.05)	(0.03)
Past production x	0.08	0.02
SNCR	(0.05)	(0.02)
Observations	310	322
R-squared	0.97	0.97

Notes: Dependent variable is observed unit level production in June-September 2003. Standard errors robust to within plant correlation are in parentheses.*Indicates significance at 5%. **Indicates significance at 1%.

Table 9: Comparing Observed Choices to a Cost-Minimizing Counterfactual

	Restructured	Regulated	Total
Estimated costs|	3.65	5.62	9.27
Observed choices	(39%)	(61%)	
($ Billion)			
Estimated costs|	3.28	3.30	6.58
Cost minimizing choices	(50%)	(50%)	
($ Billion)			

Table 10: Exposure-Based Trading Simulation Results: Facility-level decision making

	BASELINE CASE	Trading Ratio Case I (1:1.5)	Trading Ratio Case II (1:5)
High damage area	2053	1924	1596
NOx emissions (tons/day)	(55)	(78)	(146)
Low damage area	2295	2423	2750
NOx emissions (tons/day)	(55)	(78)	(146)
Total	4347	4347	4346
NOx emissions (tons/day)	(6)	(7)	(8)
% Emissions in	47%	44%	37%
High Damage Area	(1)	(1)	(3)

Notes: Standard deviations are in parentheses.

Table 11: Exposure-Based Trading Simulation Results : Company-level decision making

	BASELINE CASE	Trading Ratio Case I (1:1.5)	Trading Ratio Case II (1:5)
High damage area	2078	1930	1596
NOx emissions (tons/day)	(107)	(137)	(146)
Low damage area	2270	2418	2750
NOx emissions (tons/day)	(108)	(136)	(146)
Total	4348	4348	4346
NOx emissions (tons/day)	(10)	(7)	(8)
% Emissions in	48%	44%	37%
High Damage Area	(5)	(3)	(3)

Notes: Standard deviations are in parentheses.

Air Papers of October 18th Session
Comments by Sam Napolitano
Director, Clean Air Markets Division, U.S. Environmental Protection Agency

General

- EPA's air programs have enormous respect for the contributions that environmental and other economists have made to the Agency's efforts to better design programs. We appreciate the authors (of the air papers at this workshop) efforts to carry forward the invaluable work that economists have done over the last 35 years.
- The authors evaluate the Acid Rain Program (ARP) and the NO_x Budget Trading Program (NBP) using self-designed metrics of success. They largely ignore the reasons that Congress established for the programs. However, EPA has to set up programs under existing authorities in response to what Congress, States, and the public want done.
- The authors do not consider evidence on how well these programs have done and negatively focus on the programs not meeting objectives that the authors believe are appropriate. Reading these papers, you do not see that these programs are well designed and are highly successful at doing what they are intended to do, and more. The attached presentation that was given at the workshop on October 18th outlines air trading results and sources for program evaluations. It also provides other background information important for the authors to consider.

Shadbegian, Gray, and Morgan

- Authors briefly recognize Congress's aim for the ARP was to address an environmental issue (acid rain damage) and then focus on health benefits and costs to estimate net benefits (which Congress never intended and recent case law suggests is not allowed).
- The paper covers major aspects of the ARP -- trading vs command and control (CC) and trading ratios vs. simple trading, yet the analysis rests on an outdated air quality modeling tool (EPA has had four other better models in play since the 1996 Source-Receptor model used here was developed). Additionally, there is a very general explanation of the health and cost data and other important assumptions that leave the reader at a loss to determine if the analysis is credible. This concern is amplified when key results, such as those in Table 2, are hard to follow, presented in an inconsistent way (i.e. billions vs. millions of some year $), and appear to be partially wrong.
- The authors select ARP Phase I, which has limited value in determining overall program effectiveness and a comparative framework for CC versus trading that is different than that used when Congress made the original choice in 1990. Phase I was meant to move the trading program smoothly into place addressing the plants with the greatest sulfur dioxide emissions, but was not geared to be the final regulatory solution for these units and the rest of the power sector. Notably, Phase I was marked by limited cross-industry trading and worked through companies making internal changes to their fleets of electric generation units akin to some types of CC. Looking at ARP Phase II (coverage of entire power industry under a tighter emissions cap, where we have 6 years of experience, a lot more actual trading, and an enormous amount of emissions data available for analysis) would have provided a much better assessment. Also, it is arguable that the authors' chose the wrong comparative framework. The one stakeholders considered in 1990 when the ARP was set up would have compared allowance trading at a fixed allowance allocation level to CC achieving that level of reduction, not the level of over control reached due to the incentives provided uniquely by trading's "banking" provision. In that case, trading produces far greater net benefits than CC.
- Given the apparent simple analytics used in conjunction with the uncertainty that generally exists in this type of analysis, a very plausible conclusion is that the two approaches get roughly the same amount of benefits, but trading is cheaper than CC when the authors find that the benefits of trading and CC are within 2 percent of each other and that the trading program is close to 20 percent cheaper.

- Authors should consider framing the problem an additional way, considering that for the same cost, your analysis suggests that trading is likely to get a lot more benefits. Even at the high end of the costs per ton avoided, using scrubbers, the $94 million saved by the trading program appears to be able to provide an extra $94 million/$265 per ton = 355,000 tons of reductions. At an average value of about $15,000 per ton, those reductions would be worth over $5 billion, leaving an equal-cost trading program with several billion dollars more net benefits than CC.
- Surprised that once the authors found that the ARP Phase I had a benefit-cost ratio of about 100 to 1 they didn't point out that the overall public welfare (net benefits) could be substantially increased through regulation beyond Title IV. This analysis shows that further SO_2 controls, like EPA those provided in the Clean Air Interstate Rule (CAIR), are clearly warranted.
- Recommend replacing reference of total ARP benefits with the recently peer-reviewed article by Lauraine G. Chestnut and David Mills, *A fresh look at the benefits and costs of the US acid rain program,* Journal of Environmental Management, September 2005. It estimates the annual benefits of the ARP in 2010 at $122 billion (2000 $).

Fowlie

- The author does not recognize that the NO_x Budget Trading Program (NBP) was designed to lower ozone transport from upwind to downwind states to compliment state/local government actions to attain the 1-hour and 8-hour ozone standards. The NBP was meant to be part of a suite of federal regional measures and state/local government actions that collectively provide cost-effective control. The success of the NBP should be determined by its contribution to cost-effective ozone standard attainment, the goal that it had. Fowlie selects instead a cost-benefit framework; which the last 10 years of case law has shown Congress did not intend EPA to use.
- One of the author's two major points is that in designing the NBP, EPA did not properly factor in the differences that will occur in pollution control choices by companies that have electric generation prices that either are, or are not, regulated. She posits that due to the Averch-Johnson (A-J) effect where there is price regulation; there is a market distortion that tilts companies to use more capital intensive controls over what occurs without price regulation. Despite this contention, the author never proves that price-regulated firms chose capital-intensive controls to a greater extent than would be expected on the basis of cost-effectiveness, nor than any observed effects can be attributed to the A-J effect. In our recent examination of what occurred in states with and without price control, we found that our cost-minimization model reasonably predicts what actually has occurred under the NBP. EPA found that the more likely reasons for more capital-intensive pollution controls in price-regulated states are that there were more large units with high NO_x rates operating at higher capacity factors and facing lower construction costs as well as other factors that Fowlie did not focus on. Notably, at the time the NBP was set up EPA gave extensive consideration to the implications of the electric restructuring underway and the IPM model that EPA used was also used by FERC when it considered ways to improve restructuring due to its suitability for the task.
- There were other things going on in the last decade that further draw into question the A-J effect having a role in compliance decisions. For instance, compliance with Phase I ARP during 1995-1999 saw little, if any, of the major Southern utility power stations (where there was price regulation), select the addition of capital intensive scrubbers (they largely switched to lower sulfur coals), whereas Ellerman in 1997 reported that about half of all Phase I compliance resulted from scrubber installation.[1]
- Even if the author's point about the A-J effect was reasonable, it appears that the problem would have been created from the failure of an initiative that was supposed to provide restructuring of the power industry throughout the US, not due to poor design of the NBP per se. In 1996–1998 when EPA developed the NBP program, the Administration's position and that of many leading economists was that restructuring was occurring nationwide and the question was whether the federal government should accelerate its pace (the Clinton Administration sent Congress several bills to do

[1] Ellerman, A Denny et al, *Emissions Trading under the U.S. Acid Rain Program – Evaluation of Compliance Costs and Allowance Market Performance*, MIT CEEPR, October 1997.

so.) Notably, the market distortion that results from only partial industry restructuring after the collapse of California's system in 2000 should have been further exacerbated by the price caps many states placed on electric generation markets that are just now starting to expire. Luckily, this price control action was very substantially counterbalanced by the large economic rents received by low-cost coal-fired units, because market prices were often set by gas-fired units at the margin that were much more expensive to operate. Past analysis has shown us that even with the addition of capital-intensive pollution controls, the rents for coal-fired generation remain large so that "competition" should not lead to inordinate pressure on companies to cut capital costs. Additionally, some states actually put NBP pollution control investment in stranded asset estimates to be recovered by utilities as restructuring was phased in as an additional hedge on potential company losses of profitability. An issue that appears to have delayed, but not necessarily stopped, some cost-effective controls was the financial problems several companies had in the Northeast due to overbuilding capacity and post-Enron concerns that arose for merchant plant operators that were tarred with the brush of questionable financing. These critical aspects of restructuring are not recognized by the author while the more ephemeral A-J effect is.

- The author's second major point is that EPA should have used exposure-based trading. In a purely theoretical sense, her point is well-taken. However, some practical reflection on how to make it work shows it's likely to be problematic. Done right, there would be different trading ratios for NO_x for all the 2,600 participants in the emissions trading system that would be constantly changing as other emitters increased and decreased their emissions of NO_x and other pollutants such as VOCs that interact with NO_x to create ozone. Additionally, NO_x reduction has even greater benefits from lowering fine particle formation that should be weighed and this action also must be considered in conjunction with SO_2 emissions, if again the aim is to maximize net benefits of a program. Furthermore, one could argue that such a system should cover all sources and not just those from the power sector, if it is to truly provide the most benefits for the cheapest cost. In that case, we would have millions of sources to consider and the system would be unworkable.
- In addition, a system like this would heavily favor protection of large urban centers over rural areas. How could we explain this inequitable level of protection to Congress and the public outside of urban centers?
- There are reduced forms of exposure-based trading that could be laid out as more practical. Those companies adversely affected by these forms of trading are likely to make their application very challenging. There would be a lot of thorny technical issues such as what weather conditions should be used to develop the trading ratios (bad vs. good vs. average ozone-related years) and time period of the year (10 ozone episode used in the Mauzerall article that the author relied on vs. summer ozone season vs. annual control). In looking at this in the past, EPA has questioned whether it could be definitely assured that there would far greater benefits from such an approach that warranted the added complexity, administrative cost, potential loss of the virtual 100 percent compliance with the existing trading approaches, and added litigation burden and risk of losing litigation that would delay program implementation (and public health protection) that would occur. Notably, EPA considered simple versions of such approaches when it designed the NBP. The Agency constructed high and low NOx reduction regions that were self-contained trading regions to provide more reductions where they were most needed. Reasonable control options cost more, but did not substantially improve air quality. EPA also considered how to lower emissions from power plants contributing most to future ozone nonattainment by using trading ratios to affect such an approach. This was a lot like the simple example that Fowlie uses in her paper, but was based on much more sophisticated and detailed air quality modeling work and economic analysis.[2] EPA designed a targeted emissions reductions (TER) approach that factored in the spatial effects of ozone formation (e.g., a ton of reduction in MD was more helpful than a ton of reduction in NC to lower ozone formation in New

[2] Analysis managed by Dr. Gary Dorris was provided in a Stratus Consulting report to EPA entitled *Development and Evaluation of a Targeted Emission Reduction Scenario for NOx Point Sources in the Eastern United States: An Application of the Regional Economic Model for Air Quality (REMAQ),* November 24, 1999. This study was the outgrowth of a Phd thesis of Dr. Dorris for the same advisor for a PhD thesis that Meredith Fowlie has for this work.

York City) in an effort to provide the same air quality improvement as the simple NBP trading approach at a lower cost. The resulting approach was two-tenths of a percent cheaper than the program that EPA put in place to address the current 8-hour ozone standard without factoring in the potentially serious increases in transaction and administrative costs. There was little to show for the added complexity that would have to be introduced through trading ratios.

Hefland, Moore and Liu

- Congress authorized "banking" of allowances to increase the flexibility and cost-effectiveness of the ARP. The focus was not on finely tuned economic efficiency through temporal trading.
- EPA has seen banking as an invaluable tool in allowing the regulated community to adjust easily to changes in the economy and electric demand, leading to the power sector initially over controlling emissions and providing very large early program benefits (see first paper), and providing a glide path in the longer term for industry movement to comply with the increasingly tighter controls under the emissions caps that we first set up for SO_2 under the ARP and more recently in CAIR.
- Given that the ARP has a very active market – lots of players and a large volume of trading for today's and future allowances – and we are finding the program to be a lot less expensive with allowances prices that were much lower than expected until quite recently – and broad acceptance -- it appears to be working. Authors need to make a clearer case of about why their "theoretical findings" should mean something to those of us running the program and how we might fix "the problem."
- Note, we have found that having several pollutants in trading programs leads our linear program model (IPM) to different results from the expected "Hotelling effect" for any one pollutant. Things get a bit more complicated, as actions taken at the margin have cobenefits in addressing SO_2, NO_x, and Hg emissions.
- I recommend that the authors talk to some of the very sophisticated consultants following the market, such as ICF Resources, PEAR, Evolution Markets, NAT Source, brokers and large company trading departments – they may have much more important street wisdom to offer for why the current and futures markets behave as they do – something a 1,000 regression analyses will never reveal.
- If you are not considering how CAIR, Clean Air Mercury Rule, and the Clean Air Visibility Rule as well as New Source Review settlements that often lead to arcane allowance surrender schemes and how companies are considering the future strong possibility of mandatory carbon controls (which will at least alter, perhaps even collapse the SO_2 market), it does not appear you will ever get a handle on why this market is behaving as it does.

****

Environmental economists have made vast contributions to environmental protection over time – we have the successful air emission trading programs and advanced quantitative benefits analysis that routinely shape our major regulations. Considering further the constructive contributions that you could make in the air pollution area, I ask that you to consider working on:

- Determining in very tangible terms (like $) the benefits of protecting the environment, protecting or restoring ecological balance. Our lack of ability in this area is leading to less consideration of environmental benefits in crafting regulations.
- Where to go next on trading, identifying other sectors where we can make it work that will provide the public benefits.

Thank you for the opportunity to discuss these papers.

Discussion

Helfand et al., “Testing for Dynamic Efficiency of the Sulfur Dioxide Market”

Fowlie, “Emissions Trading, Electricity Industry Restructuring, and Investment in Pollution Abatement”

Shadbegian et al., “A Spatial Analysis of the Consequences of the SO2 Trading Program”

Nathaniel Keohane
Yale School of Management

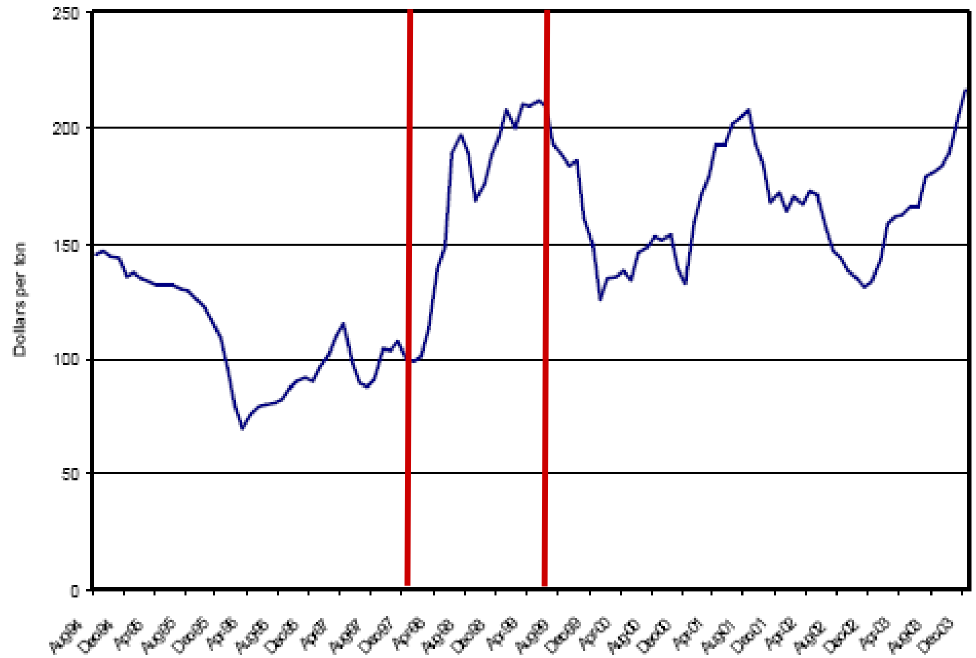
EPA Market Mechanisms and Incentives Conference
October 18, 2006

Helfand, Moore, and Liu: Overview

- Econometric analysis of SO2 allowance price movements, 1994-2003
- Two key questions:
    1. Did allowance prices follow basic Hotelling prediction?

       (No)
    2. Does information from prices in related markets (e.g., low-sulfur coal) help to explain SO2 allowance prices?

       (Yes, for wages and natural gas; no, for coal prices; still much to be explained)



Helfand, Moore, and Liu: Comments

This is an interesting (and policy-relevant) question; they bring a promising econometric method to bear; and they have a solid base of results to explore.

Three comments:

- Can more be done with the raw data?
- What do the results tell us?
- Endogeneity concerns

Helfand, Moore, and Liu: Comments

1. Can more be done with the raw data?
   - In a case like this, should be much to learn from graphical presentation of data
   - Show allowance stock (liquidity), forward market (convenience yield), etc. data in graphs
   - Summary statistics!
   - Could also show prices in other markets (natural gas, etc) alongside SO2 allowance prices

Helfand, Moore, and Liu: Comments

2. What do the results tell us?

Peculiar findings need to be explained:

- Very large and significant negative coeff on time-t price (Hotelling term)

variable	With Breaks Coef.	Std. Err.
constant	3.34	(2.90)
		[2.18]***
break1	8.21	(2.82)**
		[3.25]***
break2	-12.05	(3.31)***
		[3.41]***
$r_t^f p_t$	-13.20	(4.87)***
		[4.06]***
$(r_t^m - r_t^f)p_t$	0.05	(0.15)
		[0.19]
$elecusefe_{t+1}$	-3.07e-08	(1.45e-07)
		[1.38e-07]
$lscprcfe_{t+1}$	0.08	(0.08)
		[0.06]
$hscprcfe_{t+1}$	0.01	(0.09)
		[0.07]***
$ngasprcfe_{t+1}$	0.04	(0.01)***
		[0.01]***
$wagefe_{t+1}$	9.45	(5.84)***
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R^2	0.25	

Helfand, Moore, and Liu: Comments

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- Endogenous breaks account for a lot of the regression's fit … but even so, does not accord well with the simple data
- Low-sulfur coal price not correlated with SO2 allowance price?
- Why is wage such a strong predictor?

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Helfand, Moore, and Liu: Comments

3. Endogeneity concerns

- Econometric model:

$$p_{t+1} - p_t = \alpha + \beta_1 r_t^f p_t + \beta_2 (r_t^m - r_t^f) p_t + \varepsilon_{t+1}$$

$$\varepsilon_{t+1} = f(elecprcfe_{t+1}, lscprcfe_{t+1}, hscprcfe_{t+1}, ngasprcfe_{t+1}, wagefe_{t+1}) + v_{t+1}$$

- There seem to be clear endogeneity concerns here. Indeed, the main <u>premise</u> is that the forecast errors are related.
- Might be useful instead to think of a system of equations with a single error structure, and estimate accordingly.
- This appears to be one of the “robustness checks,” but seems to me to be central to identification.
- (Note: Would be nice to have more transparency in how price forecasting is done. Show results in appendix, specify eqns., etc.)

Fowlie: Overview

Two basic findings.

1. Power plants in states with restructured electricity markets were less likely to adopt capital-intensive compliance strategies in the lead-up to the NOx emissions trading program.

- Sophisticated econometric model of power plant compliance decisions is then used to simulate what would have happened under a counterfactual “exposure-based” trading system
  - NOx program effectively assumes “uniform mixing”, but we know that in reality, source location matters
  - A key advantage of the policy simulation is the detailed estimation of manager-specific preferences about costs

2. Exposure-based trading would have reduced emissions in high-damage areas by 6 to 22% depending on trading ratio; implies significant effects on mortality.

Fowlie: Comments

Overall, a terrific paper: frames an interesting problem, knows the data well, applies sophisticated econometric methods with care

One major comment on paper's conclusions

Fowlie: Comments

Efficiency implications for investment decisions

- Main result motivated as violating the usual assumption that all firms in the emissions market solve the same cost minimization problem
- Fowlie estimates total costs under hypothetical cost-minimizing behavior; finds that actual costs were 43% higher
- Might be useful to sort out two related issues:
  - Underlying objective function differs across firms
  - Regulation (vs. restructuring) affects investment decisions

Fowlie: Comments

Efficiency implications for investment decisions, cont'd

Are investment decisions more or less efficient in restructured markets vs. regulated markets?

- Three reasons expect more investment in regulated markets:
  - AJ effect under conventional regulation (→ overinvestment in regulated markets)
  - option value due to irreversibility and uncertainty over cost recovery in restructured states (→ less investment)
  - greater capital constraints in restructured states (→ less investment)
- But none of these says which regime is "wrong"
- Countervailing evidence in paper:
  - Firms in regulated markets ignored capital cost in their decisions
  - On other hand, approximated discount rates appear to be more reasonable in regulated markets (16% vs. 44%)

Shadbegian, Gray, and Morgan: Overview

Compares costs and benefits of Title IV SO2 trading program to two counterfactual scenarios:

1. Pre-existing regulation (weak state-level CAC regulations)
2. Uniform emissions standard to achieve observed emissions reduction

Use plant-level cost estimates along with fine-grained SR matrix to estimate benefits.

Main findings:

1. Overall net benefits were large, with benefits 100x larger than costs (benefits of $56 billion, costs of $560 million)
2. However, estimated net benefits of trading vs. CAC are <u>negative</u>; while costs were lower, benefits were also lower because plants with relatively high marginal damages emitted relatively more

Shadbegian, Gray, and Morgan: Comments

Tackles a crucial question, namely the net benefits of trading under the 1990 CAAA, and employs exactly the right benefits and (one hopes) cost data.

Two comments:

- Look more closely at substitution/compensation program?
- Simulation of trading under counterfactual policies

Shadbegian, Gray, and Morgan: Comments

1. Look more closely at substitution/compensation program?

    - Montero (1998) demonstrated the adverse selection problem inherent in voluntary “opt-in” programs
    - Those plants tended to be ones with low abatement costs.
    - Were they also plants with low marginal damages?
    - In other words, what were the net benefits from substitution and compensation?

Shadbegian, Gray, and Morgan: Comments

2. Simulation of trading under counterfactual trading-zone policies

   - Why limit trading zones to geographically contiguous areas?

   - In simulation with trading zones, market clears by scaling down allowance purchases among plants in proportion to their size
     - To the extent that abatement costs are positively correlated with marginal benefits, proportional scaling will overstate the reductions in damages achieved by trading zones
     - Seems like it would be preferable to take into account plant-level costs. (They may already have tried something like this.)

Overview

Two themes run through these papers:

1. Efficiency of real-world allowance markets

2. Emissions trading with spatial variation in marginal benefits

1. Efficiency of real-world allowance markets

Conclusions: SO2 market does not appear to have operated with full efficiency, over time or across plants.

- **HML**: Time series data on allowance prices does not support efficient markets hypothesis.
- **Fowlie**: Power plants did not make cost-effective investments under cap-and-trade program.

1. Efficiency of real-world allowance markets

Conclusions: SO2 market does not appear to have operated with full efficiency, over time or across plants.

Would be useful to draw connections to previous literature

- Work by Burtraw and Ellerman & Montero on why allowance prices were so low in Phase I
  - One reason: “Too much scrubbing”
  - Connects to both of the papers above

1. Efficiency of real-world allowance markets

Conclusions: SO2 market does not appear to have operated with full efficiency, over time or across plants.

Would be useful to draw connections to previous literature

This is also an area where anecdotal evidence from talking with folks in industry might help shed light

- A friend at Cinergy reports that his analysts thought that SO2 allowances were way underpriced at ~$200 in early 1990s
- Are there factors that industry analysts focus on that are being missed in these analyses?

1. Efficiency of real-world allowance markets

Conclusions: SO2 market does not appear to have operated with full efficiency, over time or across plants.

Would be useful to draw connections to previous literature

This is also an area where anecdotal evidence from talking with folks in industry might help shed light

And as always, we must ask: What is the relevant counterfactual?

- "Warts and all" analysis, not textbook idealization
- Especially relevant for Fowlie's analysis, since the source of variation there is in the regulation of the electricity industry, not the form of environmental policy

2. Spatial variation in marginal benefits

Conclusions: In both the NOx and SO2 markets, spatial variation in benefits matters. Emissions-based trading reduces welfare.

- **Fowlie**: Compares simulated emissions distributions under the single NOx market vs. simple exposure-based trading.
- **SGM**: Estimate welfare consequences of a single SO2 market, compared with a command-and-control counterfactual

2. Spatial variation in marginal benefits

<u>Conclusions</u>: In both the NOx and SO2 markets, spatial variation in benefits matters. Emissions-based trading reduces welfare.

What is the magnitude of the effect?

At first reading, Fowlie and SGM have very different takes on effectiveness of simple trading rules

- Fowlie: Simple geographic trading rules make a big difference
- SGM: Simple geographic trading rules don’t make much difference

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- Fowlie: Simple geographic trading rules make a big difference
- SGM: Simple geographic trading rules don't make much difference

In fact, the difference is smaller than it might appear

- Fowlie: 6-22% difference in emissions in high-damage areas
- SGM: 10-14% decrease in damages (increase in benefits)

2. Spatial variation in marginal benefits

Conclusions: In both the NOx and SO2 markets, spatial variation in benefits matters. Emissions-based trading reduces welfare.

What is the magnitude of the effect?

What are the alternatives?

- Trading ratios or transfer prices based on relative impacts
- SGM have the information needed to do this in principle
- Indeed, with the SR matrix in hand it is much harder to do the efficiency analysis than to design an efficient policy instrument
  - The latter does not require information on plant-level costs
- Here we bump up against the science
  - Atmospheric chemists complain about even the PM10 SR matrix
  - Modeling ozone precursors such as NOx appears very hard

Market Mechanisms and Incentives: Applications to Environmental Policy

A Workshop Sponsored by the U.S. Environmental Protection Agency's National Center for Environmental Economics (NCEE) and National Center for Environmental Research (NCER)

Resources for the Future
1616 P Street, NW, Washington, DC 20036
October 17-18, 2006

Disclaimer

Market Mechanisms and Incentives: Applications to Environmental Policy

Resources for the Future
1616 P Street, NW, Washington, DC 20036
(202) 328-5000

October 17th – 18th, 2006

October 17, 2006: Market Mechanisms in Environmental Policy

8:00 a.m. – 8:45 a.m. **Registration**

8:45 a.m. – 11:45 a.m. **Session I: Brownfields and Land Issues**
Session Moderator: **Robin Jenkins**, EPA, National Center for Environmental Economics

8:45 a.m. – 9:00 a.m. Introductory Remarks: **Sven-Erik Kaiser**, EPA, Office of Brownfields Cleanup and Redevelopment

9:00 a.m. – 9:30 a.m. Environmental Liability and Redevelopment of Old Industrial Land
Hilary Sigman, Rutgers University

9:30 a.m. – 10:00 a.m. Incentives for Brownfield Redevelopment: Model and Simulation
Peter Schwarz and **Alex Hanning**, University of North Carolina at Charlotte

10:00 a.m. – 10:15 a.m. **Break**

10:15 a.m. – 10:45 a.m. Brownfield Redevelopment Under the Threat of Bankruptcy
Joel Corona, EPA, Office of Water, and **Kathleen Segerson**, University of Connecticut

10:45 a.m. – 11:00 a.m. Discussant: **David Simpson**, EPA, National Center for Environmental Economics

11:00 a.m. – 11:15 a.m. Discussant: **Anna Alberini**, University of Maryland

11:15 a.m. – 11:45 a.m. Questions and Discussion

11:45 a.m. – 12:45 p.m. **Lunch**

12:45 p.m. – 2:45 p.m. **Session II: New Designs for Incentive-Based Mechanisms for Controlling Air Pollution**
Session Moderator: **Will Wheeler**, EPA, National Center for Economic Research

12:45 p.m. – 1:15 p.m. Dynamic Adjustment to Incentive-Based Environmental Policy To Improve Efficiency and Performance
Dallas Burtraw, Danny Kahn, and Karen Palmer, Resources for the Future

	1:15 p.m. – 1:45 p.m.	Output-Based Allocation of Emissions Permits for Mitigating Tax and Trade Interactions **Carolyn Fischer**, Resources for the Future
	1:45 p.m. – 2:00 p.m.	Discussant: **Ann Wolverton**, EPA, National Center for Environmental Economics
	2:00 p.m. – 2:15 p.m.	Discussant: **Arik Levinson**, Georgetown University
	2:15 p.m. – 2:45 p.m.	Questions and Discussion
2:45 p.m. – 3:00 p.m.	**Break**	
3:00 p.m. – 5:30 p.m.	**Session III: Mobile Sources** Session Moderator: **Elizabeth Kopits**, EPA, National Center for Environmental Economics	
	3:00 p.m. – 3:30 p.m.	Tradable Fuel Economy Credits: Competition and Oligopoly **Jonathan Rubin**, University of Maine; **Paul Leiby**, Environmental Sciences Division, Oak Ridge National Laboratory; and **David Greene**, Oak Ridge National Laboratory
	3:30 p.m. – 4:00 p.m.	Do Eco-Communication Strategies Reduce Energy Use and Emissions from Light Duty Vehicles? **Mario Teisl**, **Jonathan Rubin**, and **Caroline L. Noblet**, University of Maine
	4:00 p.m. – 4:30 p.m.	Vehicle Choices, Miles Driven, and Pollution Policies **Don Fullerton**, **Ye Feng**, and **Li Gan**, University of Texas at Austin
	4:30 p.m. – 4:45 p.m.	Discussant: **Ed Coe**, EPA, Office of Transportation and Air Quality
	4:45 p.m. – 5:00 p.m.	Discussant: **Winston Harrington**, Resources for the Future
	5:00 p.m. – 5:30 p.m.	Questions and Discussion
5:30 p.m.	**Adjournment**	

October 18, 2006:

8:45 a.m. – 9:15 a.m.	**Registration**	
9:15 a.m. – 12:20 p.m.	**Session IV: Air Issues** Session Moderator: **Elaine Frey**, EPA, National Center for Environmental Economics	
	9:15 a.m. – 9:45 a.m.	Testing for Dynamic Efficiency of the Sulfur Dioxide Allowance Market **Gloria Helfand**, **Michael Moore**, and **Yimin Liu**, University of Michigan
	9:45 a.m. – 10:05 a.m.	When To Pollute, When To Abate: Evidence on Intertemporal Use of Pollution Permits in the Los Angeles NOx Market **Michael Moore** and **Stephen P. Holland**, University of Michigan

10:05 a.m. – 10:20 a.m.	**Break**	
	10:20 a.m. – 10:50 a.m.	A Spatial Analysis of the Consequences of the SO_2 Trading Program **Ron Shadbegian**, University of Massachusetts at Dartmouth; Wayne Gray, Clark University; and Cynthia Morgan, EPA
	10:50 a.m. – 11:20 a.m.	Emissions Trading, Electricity Industry Restructuring, and Investment in Pollution Abatement **Meredith Fowlie**, University of Michigan
	11:20 a.m. – 11:35 a.m.	Discussant: **Sam Napolitano**, EPA, Clean Air Markets Division
	11:35 a.m. – 11:50 a.m.	Discussant: **Nat Keohane**, Yale University
	11:50 a.m. – 12:20 p.m.	Questions and Discussion
12:20 p.m. – 1:30 p.m.	**Lunch**	
1:30 p.m. – 4:35 p.m.	**Session V: Water Issues** Session Moderator: **Cynthia Morgan**, EPA, National Center for Environmental Economics	
	1:30 p.m. – 2:00 p.m.	An Experimental Exploration of Voluntary Mechanisms to Reduce Non-Point Source Water Pollution With a Background Threat of Regulation Jordan Suter, Cornell University, Kathleen Segerson, University of Connecticut, Christian Vossler, University of Tennessee, and Greg Poe, Cornell University
	2:00 p.m. – 2:30 p.m.	Choice Experiments to Assess Farmers' Willingness to Participate in a Water Quality Trading Market **Jeff Peterson**, Washington State University, and Sean Fox, John Leatherman, and Craig Smith, Kansas State University
2:30 p.m. – 2:45 p.m.	**Break**	
	2:45 p.m. – 3:15 p.m.	Incorporating Wetlands in Water Quality Trading Programs: Economic and Ecological Considerations **Hale Thurston** and Matthew Heberling, EPA, National Risk Management Research Laboratory, Cincinnati, Ohio
	3:15 p.m. – 3:35 p.m.	Designing Incentives for Private Maintenance and Restoration of Coastal Wetlands **Richard Kazmierczak** and **Walter Keithly**, Louisiana State University at Baton Rouge
	3:35 p.m. – 3:50 p.m.	Discussant: **Marc Ribaudo**, USDA, Economic Research Service
	3:50 p.m. – 4:05 p.m.	Discussant: **Jim Shortle**, Pennsylvania State University
	4:05 p.m. – 4:35 p.m.	Questions and Discussions
4:35 p.m. – 4:45 p.m.	**Final Remarks**	

4:45 p.m. **Adjournment**

Voluntary-Threat Mechanisms to Reduce Ambient Water Pollution

Jordan F. Suter[1], Kathleen Segerson[3], Christian A. Vossler[2], and Gregory L. Poe[1]

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[2] Department of Economics, University of Tennessee, Knoxville, TN 37996
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Voluntary-Threat Mechanisms to Reduce Ambient Water Pollution

Abstract: Given the political and economic attractiveness of addressing nonpoint source water pollution through a voluntary mechanism that carries with it a background threat of a mandatory policy (e.g. an ambient tax), this paper expands on recent theoretical work by Segerson and Wu (*Journal of Environmental Economics and Management*, 2006) in two important ways. First, we suggest a modification of the theory that generates optimal, voluntary abatement as part of a subgame perfect Nash equilibrium without the necessity for applying taxes retroactively. Second, we use laboratory economic experiments to test the voluntary/threat policy suggested by Segerson and Wu as well as the policy that we introduce, and compare them to a pure ambient tax policy. Our experimental results indicate that the voluntary/threat policy behaves as well or better than the pure tax policy, though these outcomes are highly dependent on the form and parameters of the mechanism.

Keywords: voluntary mechanisms; ambient-based tax; nonpoint source pollution; laboratory experiments

I. Introduction

Improvements in surface water quality since the passage of the Federal Clean Water Act Amendments of 1972 have come primarily as a result of reductions in emissions from point sources, such as wastewater treatment plants and factories. While opportunities for further reduce emissions from point sources remain, it is nonpoint source pollution that presently represents the greatest share of surface water impairment in the United States (Ribaudo 2003). Agricultural production, which occurs on approximately 60% of nonfederal land in the US (NRI 2002), is the largest component of nonpoint source water pollution and represents the leading source of water quality impairments among the rivers and lakes surveyed in the 2000 *National Water Quality Inventory* (US EPA 2002).

Given the role of nonpoint sources in influencing water quality, economic theorists have devised a number of mandatory approaches designed to reduce surface water pollution stemming from agricultural production. These approaches can be roughly broken into performance-based policies, which base regulation on measurable outcomes, and design-based policies, which are predicated on input and land management decisions (Ribaudo 1999). Since nonpoint source emissions are characterized as prohibitively costly to monitor on a firm-level basis, performance-based policies have been directed towards ambient environmental conditions. Beginning with the seminal work of Segerson (1988), numerous mandatory approaches that provide incentives to nonpoint source polluters based on ambient pollution levels have been proposed (e.g., Xepapadeas 1991; Cabe and Herriges 1992; Hansen 1998; Horan et al. 1998; Karp 2004).

One main criticism of mandatory approaches, in particular policies that involve taxing nonpoint polluters based on ambient pollution, is political feasibility. Policy makers have historically addressed nonpoint source pollution almost exclusively through voluntary measures[1]. While voluntary programs have been widely accepted by agricultural producers, there is little evidence that they have delivered outcomes, in terms of improved water quality, that would warrant declaring them a success (Shortle, Abler and Ribaudo 2001). A recent study by the US Environmental Protection Agency (EPA) finds that almost 35 years after the passage of the Clean Water Act Amendments, nearly 70% of all stream miles in the United States can be classified as being in "fair" or "poor" condition (US EPA 2006).

In an effort to wed the political palatability of a voluntary policy with the theoretical attractiveness of an appropriately designed mandatory policy, Segerson and Wu (2006) introduce a policy that uses voluntary and mandatory programs as complementary instruments. The proposed policy allows firms in a watershed to voluntarily meet an ambient pollution standard. As long as the ambient standard is achieved, no regulatory fees are charged. If, however, the standard is not met voluntarily, then a mandatory instrument is put in place, in particular an ambient tax policy. The threatened tax policy is structured in such a way that firms are induced to meet the ambient pollution standard voluntarily.

The proposed voluntary/threat policy has some clear advantages over a strictly mandatory or strictly voluntary approach. From a producer's standpoint the policy is attractive because it allows for flexibility in meeting pollution standards without explicit regulation. From the regulator's standpoint the policy's attractiveness comes from avoiding the potentially large costs associated with administering the tax and incurring the information costs necessary to appropriately set the tax rate. Finally, the instrument is attractive from the social planner's perspective, as it offers the potential to cost effectively address the nonpoint source pollution problem.

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[1] Common voluntary policies include land retirement programs, such as the Conservation Reserve Program, as well as working land programs, that provide incentives to agricultural landowners for developing best management practices (BMPs) and implementing pollution prevention and control measures. Annual federal expenditures for voluntary conservation programs are projected to be nearly $5 billion by 2011 (ERS 2002)

A shortcoming of the Segerson and Wu framework is a coordination problem in the voluntary setting resulting from the existence of multiple Nash Equilibria, including a possible equilibrium where no firm engages in pollution control. To eliminate the multiplicity of Nash Equilibria, we modify the threatened tax mechanism such that, if a violation occurs, expected tax payments under the subsequent tax policy are a function of the extent of the voluntary period violation. The threat mechanism can be parameterized in ways that leave optimal abatement in the voluntary setting the unique equilibrium.

Due to the novelty of a voluntary/threat policy, empirical program evaluation using naturally occurring data is difficult[2] since no such program is presently being implemented. The potential social gains from firms voluntary achieving a pollution standard at least-cost, together with the theoretic potential for socially suboptimal behavior imply that the experimental economics laboratory is an important alternative testing ground for gaining a comparative perspective of how the proposed policies will work in practice.

In recent years a burgeoning set of studies have complemented the theoretical literature by testing many of the proposed regulatory policies in an experimental economics laboratory setting (Spraggon 2002, 2004; Alpizar et al. 2004; Poe et al. 2004, Cochard et al. 2005, Suter et al. 2006, Vossler et al. 2006). The results from these experimental studies show that a subset of the proposed theoretical policies, including a tax policy similar to the threatened policy of Segerson and Wu, engenders outcomes that are highly efficient.

In the next section of the paper we provide a theoretical background for the policy introduced by Segerson and Wu and the endogenous policy that we propose. In Section III we explain the experimental design and outline five hypotheses to be tested. In Section IV we present and analyze the experimental results and then conclude the paper in Section V with a summary of our findings and a discussion of their policy relevance.

[2] Compliance mechanisms require farmers to undertake conservation measures to be eligible for some Federal aid programs. For example farmers that fail to reduce soil erosion on highly erodible land may be ineligible for some Federal benefits (USDA 2004). While this is similar to a voluntary/threat policy, the threat is based more on input decisions than on the actual effluent generated.

II. Theoretical Background

Our model follows that of Segerson and Wu closely.[3] Suppose there are n firms, denoted by i, in a given watershed. Let a_i denote abatement, and $C_i = C(a_i, \theta_i)$ the abatement cost function where θ_i is an index that represents characteristics specific to the firm.. We assume that the cost function is strictly convex, with $C'(a_i, \theta_i) > 0, C''(a_i, \theta_i) > 0$ and $C(0, \theta_i) = 0$. Ambient pollution at a monitoring point, denoted by x, is a function of the abatement decisions of all firms, with $x = x(a_1, \ldots a_n; \theta_1, \ldots, \theta_n)$, with $x'(a_i, \theta_i) < 0$ and $x''(a_i, \theta_i) \geq 0$. Given that abatement is costly, in the absence of any policy intervention we expect $a_i = 0$.

Now suppose that a social planner is interested in reducing ambient pollution to an exogenously determined water quality standard, which we denote x^s. The standard could be based on a Total Maximum Daily Load (TMDL) requirement or simply be a product of political bargaining. The social planner's problem and corresponding Lagrangian, assuming an interior solution, can then be written as [4]

$$\underset{a_i}{Min} \sum_{i=1}^{n} C(a_i, \theta_i) \quad s.t. \;\; x(a_1, \ldots, a_n; \theta_1, \ldots, \theta_n) \leq x^s, \;\; a_i \geq 0 \tag{1}$$

$$L = -\sum_{i=1}^{n} C(a_i, \theta_i) + \lambda(x^s - x(\cdot)). \tag{2}$$

The strict monotinicity and convexity of the firm cost functions imply that the first-order conditions are solved with $\lambda^* = -C'(a_i^*)/x'(a_i^*)$ for all i. The λ term can be interpreted as the marginal benefit to firms of increasing the ambient standard by one unit. Since $\lambda^* > 0$, the constraint is binding and therefore ambient pollution is exactly equal to the standard.

In the following subsections we detail the theoretical basis for three policies that seek to induce polluters in a watershed to achieve the ambient pollution standard at least cost. The first case, a pure ambient tax policy, and the second case, a voluntary policy with a threat of an exogenously determined tax, are very similar to those described by Segerson and Wu and therefore we do not provide formal proofs. We treat the third case, a voluntary policy with a

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[3] In particular, we make the assumptions that abatement and firm characteristics can each be represented by a scalar, and that the policy goal is one of meeting an ambient water quality standard on average such that stochastic factors (e.g. weather) can presumably be suppressed.

[4] An interior solution implies that the relevant case has $a_i^* > 0$ for all i. If this were not true, then the regulator would be unnecessarily exposing one or more firms to potential tax liabilities when the firm(s) should clearly not be regulated.

threat of an endogenously determined tax, more rigorously and provide several proofs to establish the policy's theoretical properties.

IIa. Tax Policy

Suppose that the social planner is attempting to reach the ambient standard at least cost through the use of a policy that charges all firms in the watershed a marginal tax, τ, on units of ambient pollution above an ambient standard. Defining the tax rate $\tau = \lambda^*$, and using the superscript t to indicate abatement under the tax policy, the cost minimization problem for firm i is

$$\underset{a_i^t}{Min} \quad C\left(a_i^t, \theta_i\right) + max\left(0, \tau \cdot \left(x\left(a_1, \ldots a_n; \theta_i, \ldots \theta_n\right) - x^s\right)\right) \tag{3}$$

Under the tax policy $\boldsymbol{a}^t = \boldsymbol{a}^*$ is a Nash Equilibrium[5] (NE), where a_i^* is the cost minimizing abatement level for firm i as defined previously. To show this, suppose the $n-1$ firms choose abatement level $a_i^t = a_i^*$. Recalling that $\tau = \lambda^*$, every unit of abatement by firm i that is less than a_i^* will cost the firm $-C'\left(a_i^*\right)/x'\left(a_i^*\right)$ in terms of tax payments, while the per unit abatement costs avoided will be less than $-C'\left(a_i^*\right)/x'\left(a_i^*\right)$, since $C''\left(a_i\right) < 0$ and $x''\left(a_i\right) \geq 0$. Therefore firm i will be strictly worse off by choosing $a_i^t < a_i^*$ as opposed to $a_i^t = a_i^*$. Firm i will also be strictly worse off by choosing $a_i^t > a_i^*$, since abating to the point where ambient pollution is below the ambient standard is more costly to the firm than meeting the standard with equality and there is no benefit in terms of additional tax penalties avoided.

Further, $\boldsymbol{a}^t = \boldsymbol{a}^*$ is a unique NE since a firm choosing $a_i^t > a_i^*$ would incur per unit abatement costs in excess of the tax rate, τ, and could therefore never be optimal. Given that none of the n firms choose $a_i^t > a_i^*$, no firm will rationally choose $a_i^t < a_i^*$, since this would result in pollution in excess of the ambient standard.

An important feature of the tax policy is that X^s in equation (3) can be replaced with a tax threshold, $\bar{x} \leq x^s$ such that the unique NE $\boldsymbol{a}^t = \boldsymbol{a}^*$ is maintained. Therefore $\bar{x}$ is a choice variable and setting $\bar{x}$ below x^s has the effect of increasing tax payments, while $a_i^t = a_i^*$ remains

[5] ***Bold*** typeface is used to signify a vector.

an optimal choice for each of the n firms. We can therefore define the cost to each firm of one period of the tax policy as

$$C(a_i^*, \theta_i) + \tau(x^s - \bar{x}). \tag{4}$$

IIb. Voluntary Policy with Exogenous Threat Mechanism

If the policy maker allows firms to meet the pollution standard voluntarily without incentives, we expect firms to expend zero abatement effort, since abatement is expensive. However, now suppose that the policy maker allows firms to respond to the pollution standard voluntary, but includes a threat of a tax policy if the standard is not achieved. Specifically, if ambient pollution is above the standard then the tax policy described above is put into place for $K \leq \infty$ periods.[6] The key parameter in the voluntary-threat mechanism is the pollution threshold, as the incentive to meet the standard voluntarily is provided by employing a tax threshold that is significantly less than the standard, which makes the tax payments – even under optimal abatement in the tax stage game – strictly positive for each firm. We label this as an "exogenous" threat mechanism as the pollution threshold is exogenous in the sense that it does not depend on behavior in the voluntary period in which a violation occurred.

Incorporating the superscript v to indicate outcomes in the voluntary stage, the amount of voluntary abatement chosen by firm i is denoted a_i^v. We further define $a_i^{v(s)}$ as the amount of voluntary abatement necessary by firm i to ensure that the ambient standard is exactly met, given the abatement activities of the other firms in the watershed, such that $x(a_i^{v(s)}, a_{-i}^v; \theta_i, \theta_{-i}) = x^s$. For the ambient standard to be met voluntarily, the threatened tax policy must be sufficiently costly so that the standard is met in the voluntary stage. We have already shown that in every period of the tax stage firms will choose a_i^* as part of a unique NE when $\tau = -C'(a_i^*) / x'(a_i^*)$. Next we show that if the costs imposed by the tax policy are sufficiently high, then each firm will choose $a_i^v = a_i^{v(s)}$ in the voluntary stage game as part of a subgame perfect NE (SPNE).

[6] Allowing the possibility that K is finite is a trivial variation on Segerson and Wu, who assume that the tax policy is imposed in perpetuity. Considering the finite case is important for purposes of experimental testing. In particular, it allows us to end and re-start the game in experimental sessions where a violation occurs, akin to a situation where the regulator gives firms a second chance to comply voluntarily.

It must be the case that firm *i* will optimally choose either $a_i^v = a_i^{v(s)}$ or $a_i^v = 0$ in the voluntary stage. It is never optimal for a firm to choose $a_i^v > a_i^{v(s)}$ since this results in ambient pollution that is strictly less than the standard and firm *i* would be better off choosing $a_i^v = a_i^{v(s)}$ so that the standard is achieved with equality. It is also never optimal for firm *i* to choose $0 < a_i^v < a_i^{v(s)}$. If $a_i^v < a_i^{v(s)}$ then the ambient standard will not be met, the tax policy will be imposed and firm *i* should not choose a positive level of abatement, since abatement is costly.

In the voluntary period the firm therefore has a choice between abating so that the ambient standard is achieved or not abating at all and paying the tax over the next *K* periods. Assuming a discount factor 0 < δ < 1, the cost of voluntary abatement sufficient to meet the standard across *K*+1 periods is given by $\sum_{k=0}^{K} \delta^k C\left(a_i^{v(s)}, \theta_i\right)$. The cost of abating zero in the voluntary period and facing *K* periods of the tax policy, is given by $\sum_{k=1}^{K} \delta^k \left(C\left(a_i^*, \theta_i\right) + \tau\left(x^s - \bar{x}\right)\right)$. Therefore a firm will abate voluntarily, and the standard will be achieved, if

$$\sum_{k=0}^{K} \delta^k C\left(a_i^{v(s)}, \theta_i\right) \leq \sum_{k=1}^{K} \delta^k \left(C\left(a_i^*, \theta_i\right) + \tau\left(x^s - \bar{x}\right)\right). \tag{5}$$

When $\bar{x} = x^s$, the expected liabilities are zero under the tax policy. In this case, no firm would ever choose $a_i^v \geq a_i^*$, since $\mathrm{C}\left(\mathrm{a}_\mathrm{i}^\mathrm{v}, \theta_i\right) \geq \mathrm{C}\left(\mathrm{a}_\mathrm{i}^*, \theta_i\right)$. Therefore the standard will not be achieved voluntarily and each firm is strictly better off by choosing $a_i^v = 0$.

When $\bar{x}$ is sufficiently below x^s, there is a SPNE whereby each firm chooses abatement strategy $a_i^v = a_i^{v(s)} = a_i^*$ in the voluntary stage and $a_i^v = a_i^*$ in the tax stage. To see this, suppose $\boldsymbol{a}_{-i}^v = \boldsymbol{a}_{-i}^*$ so that $a_i^{v(s)} = a_i^*$ for firm *i*. Recall that firm *i* will either choose $a_i^v = a_i^{v(s)}$ or $a_i^v = 0$. Choosing $a_i^v = 0$ will result in the standard not being met and the imposition of the tax policy. Therefore if $C\left(a_i^*, \theta_i\right) \leq \sum_{k=1}^{K} \delta^k \tau\left(x^s - \bar{x}\right)$ then firm *i* will optimally choose $a_i^v = a_i^{v(s)} = a_i^*$. Note that in order for $\boldsymbol{a}^v = \boldsymbol{a}^*$ to be part of a SPNE, $\bar{x}$ must be chosen so that $C\left(a_i^*, \theta_i\right) \leq \sum_{k=1}^{K} \delta^k \tau\left(x^s - \bar{x}\right)$ for each firm.

Under the voluntary-threat policy introduced by Segerson and Wu there will also exist SPNE whereby the ambient standard is achieved at greater than least cost unless

$C\left(a_i^*, \theta_i\right) = \sum_{k=1}^{K} \delta^k \tau\left(x^s - \bar{x}\right)$ for each of the n firms. If not, then firms for which this holds with inequality would have an incentive to overabate as they strictly prefer the voluntary policy to the tax policy. For this condition to hold with equality would require that all firms have identical abatement costs at the optimum and that $\bar{x} = x^s - \left(C\left(a_i^*, \theta_i\right) \Big/ \sum_{k=1}^{K} \delta^k \tau \right)$. As an example of a SPNE whereby the ambient standard is achieved voluntarily at greater than least cost, suppose that $\boldsymbol{a}_{-i}^v < \boldsymbol{a}_{-i}^*$ so that in order for the standard to be met, firm i must choose $a_i^v = a_i^{v(s)} = a_i^* + \varepsilon$. This will be an optimal choice for firm i if $\sum_{k=0}^{K} \delta^k C\left(a_i^* + \varepsilon, \theta_i\right) < \sum_{k=1}^{K} \delta^k \left[C\left(a_i^*, \theta_i\right) + \tau\left(x^s - \bar{x}\right)\right]$ and otherwise it will optimally choose $a_i^v = 0$. Therefore, for all values of

$$\bar{x} < x^s - \frac{C\left(a_i^* + \varepsilon, \theta_i\right) + \sum_{k=1}^{K} \delta^k \left[C\left(a_i^* + \varepsilon, \theta_i\right) - C\left(a_i^*, \theta_i\right)\right]}{\sum_{k=1}^{K} \delta^k \tau}$$ firm i will optimally choose

$a_i^v = a_i^{v(s)} = a_i^* + \varepsilon$ and the ambient standard will be met voluntarily. None of the other firms have an incentive to deviate from their strategy since any increase in abatement effort will impose costs without a reduction in liabilities and any decrease in abatement effort will result in the standard not being met and the consequent costs of the tax policy being greater than the savings in abatement costs.

When $a_i^{v(s)} \neq a_i^*$ for at least one of the firms in the watershed, the costs of meeting the ambient standard are not minimized. As $\bar{x}$ diverges from x^s the range of optimal voluntary abatement levels expands and the potential for free riding increases. This implies a tradeoff in the choice of the tax threshold. Setting $\bar{x}$ low relative to x^s generates a more draconian incentive for firms to meet the standard voluntarily, but opens the door to greater disparities between optimal and realized abatement choices.

In addition to multiple SPNE where the ambient standard is achieved voluntarily, there also is a SPNE whereby all firms choose zero abatement in the voluntary period. If $\boldsymbol{a}_{-i}^v = \mathbf{0}$, firm i will also choose to abate zero units since abating to the point where the ambient standard is met is excessively costly or not feasible[7]. In past experimental analyses of ambient regulatory policies with a zero abatement NE, in addition to the pareto optimal NE, groups achieved significantly lower levels of social efficiency than under the ambient policies that did not have a zero abatement NE (Spraggon 2002, Vossler et al. 2006). Unfortunately, in the case of the voluntary policy with the exogenous threat the choice of $\bar{x}$ alone cannot eliminate the zero abatement NE.

IIc. Voluntary Policy with Endogenous Threat Mechanism

To eliminate the existence of the suboptimal equilibria in the voluntary stage, Segerson and Wu suggest the threat of a retroactive tax policy. Under this policy, if the ambient standard is violated in the voluntarily stage, firms pay taxes for the violation in the voluntary stage, in addition to facing the tax policy in future periods The tax paid on ambient pollution in excess of the standard in the voluntary stage would be collected prior to the first round of the tax stage. While this does eliminate the zero abatement outcome in the voluntary stage, it seems to negate much of the political attractiveness associated with the voluntary policy. The voluntary policy with a retroactive tax distinguishes itself from a pure tax policy only in the sense that rather than being collected at the end of the period, taxes in the voluntary stage are collected at the beginning of the next period.

Retaining the flavor of the retroactive tax, we introduce a new policy instrument where the tax threshold is endogenously determined. In particular, the threshold in the tax stage is determined by the level of noncompliance in the voluntary stage. Therefore under voluntary noncompliance, this instrument makes the amount of future tax bills conditional on voluntary period behavior. This implies that even if all other firms undertake zero abatement, for example, firm *i* has an incentive to abate to reduce future tax payments. Formally, if the ambient standard is exceeded in the voluntary stage, then the tax payment due in each round of the tax stage is defined as

$$\text{Tax Payment} = \tau\left[x^t - \tilde{x}\right] \;\; where\, \tilde{x} = x^s - \varphi\left(x^v - x^s\right) \; and \;\; \varphi > 0 . \qquad (6)$$

[7] There is a potential that the best response for firm *i* would be to meet the standard voluntarily even if all other firms chose zero abatement, however this would require that the standard be relatively close to the baseline level of ambient pollution and is therefore not of particular interest.

The scaling parameter, φ, is freely chosen by the regulator. Increasing φ lowers the tax threshold for all levels of $x^v > x^s$, where x^v denotes realized pollution in the voluntary stage, and therefore increases the severity of the threatened tax policy. The crux of the mechanism is that the tax threshold decreases as the level of pollution in excess of the standard in the voluntary stage increases, thus making the consequent tax policy more costly to firms. The tax payment in each period of the tax stage can then be written as $\tau\left[x^t + \varphi x^v - (1+\varphi)x^s\right]$ for pollution levels greater than $\widetilde{x}$. From this representation, it is apparent that in the tax stage of the endogenous mechanism, firms pay a tax based on the pollution levels in that period as well as a scaled tax on the pollution that occurred in the voluntary stage.

We have shown that in the tax stage any threshold, $\bar{x} \leq x^s$ will induce a unique NE $\boldsymbol{a}^t = \boldsymbol{a}^*$, which implies that the standard is met at least cost. Simplifying equation (6) and multiplying it by the discount rate yields the tax penalty over K rounds from voluntary noncompliance under the endogenous threat mechanism

$$\sum_{k=1}^{K} \delta^k \tau\varphi\left(x^v - x^s\right). \tag{7}$$

In the voluntary stage, each firm compares the cost of abatement against the discounted stream of future tax payments, however the severity of the penalty is now a function of each firms' voluntary abatement decision. The result is the potential elimination of suboptimal equilibria. The equilibrium conditions generated by the endogenous tax threat are derived in Propositions 1 and 2 below.

Proposition 1: *If* $\tau = \lambda^*$ *then* $\{\boldsymbol{a}^v, \boldsymbol{a}^t\} = \{\boldsymbol{a}^*, \boldsymbol{a}^*\}$ *is a unique SPNE if and only if* $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$.

Proof of Proposition 1: We have already shown that in the tax stage the strategy $\boldsymbol{a}^t = \boldsymbol{a}^*$ is a unique NE. In proving Proposition 1 we start by showing that when $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$ and $\boldsymbol{a}^v_{-i} = \boldsymbol{a}^*_{-i}$ then firm i's best response is to choose $a^v_i = a^*_i$. In the second part of the proof we show that when $\varphi < \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$ and $\boldsymbol{a}^v_{-i} = \boldsymbol{a}^*_{-i}$, then it is not a best response for firm i to choose $a^v_i = a^*_i$.

Finally in the third part, we show that as long as $\varphi \geq \left(\sum_{k=1}^{K} \delta^k \right)^{-1}$ then $\boldsymbol{a}^v = \boldsymbol{a}^*$ can be the only SPNE.

When $\boldsymbol{a}^v_{-i} = \boldsymbol{a}^*_{-i}$, then the standard will be achieved exactly if firm *i* chooses $a_i^v = a_i^*$. In this case the tax policy will not be imposed and the cost to the firm over *K*+1 periods will be $\sum_{k=0}^{K} \delta^k C(a_i^*, \theta_i)$. If firm *i* chooses $a_i^v > a_i^*$ then ambient pollution will be below the standard and the tax policy will again not be imposed. The cost of choosing $a_i^v > a_i^*$ is greater than the cost of choosing $a_i^v = a_i^*$, however, which implies that this is not a best response. This result does not depend on the choice of φ.

If firm *i* chooses $a_i^v < a_i^*$ then ambient pollution will exceed the standard and the tax policy will be put into place. The cost of the tax policy will depend on the firm's voluntary abatement decision. Firm *i's* optimal choice of voluntary abatement at or below a_i^* can be represented by the minimization problem

$$\underset{a_i^v}{Min} \quad C(a_i^v, \theta_i) + \sum_{k=1}^{K} \delta^k \left[C(a_i^*, \theta_i) + \tau\varphi\left(x(a_i^v, \theta_i) - x(a_i^*, \theta_i) \right) \right] \text{ s.t. } \mathrm{a}_\mathrm{i}^\mathrm{v} \leq a_i^* . \tag{8}$$

The Kuhn-Tucker conditions associated with equation (8) are

$$C'\left(a_i^v\right) + \tau\varphi \sum_{k=1}^{K} \delta^k x'\left(a_i^v\right) + \mu = 0 \tag{8a}$$

$$\mu\left(a_i^* - a_i^v\right) = 0 \tag{8b}$$

$$\mu \geq 0 . \tag{8c}$$

We have to consider two possible solutions, one with $\mu \geq 0$ and $a_i^v = a_i^*$ and the other with $a_i^v < a_i^*$ and $\mu = 0$. Recall that $\tau = -C'\left(a_i^*\right) / x'\left(a_i^*\right)$ so that when $\varphi \geq \left(\sum_{\mathrm{k}=1}^{\mathrm{K}} \delta^{\mathrm{k}} \right)^{-1}$ then condition (8a) implies $C'\left(a_i^v\right) + \mu \geq C'\left(a_i^*\right) \cdot \dfrac{x'\left(a_i^v\right)}{x'\left(a_i^*\right)}$. Clearly, the former solution will hold, since $x''\left(a_i\right) \geq 0$ and therefore it follows that $C'\left(a_i^v\right) + \mu \geq C'\left(a_i^*\right)$ for all values of $\mu \geq 0$. The latter solution, however, implies $C'\left(a_i^v\right) \geq C'\left(a_i^*\right)$, which cannot be true given the strict convexity of the cost function. So firm *i* minimizes costs given that the tax policy will be put in place by choosing

$a_i^v = a_i^*$, however, this represents the situation whereby the standard is met. The costs of choosing $a_i^v < a_i^*$ are therefore always greater than the cost of $a_i^v = a_i^*$, thus when $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$ and $\boldsymbol{a}_{-i}^v = \boldsymbol{a}_{-i}^*$, the unique best response for firm *i* is to choose $a_i^v = a_i^*$.

When $\varphi < \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$, then $C'\left(a_i^v\right) + \mu < C'\left(a_i^*\right) \cdot \frac{x'\left(a_i^v\right)}{x'\left(a_i^*\right)}$, in which case the K-T conditions (8a)-(8c) are solved only with $a_i^v < a_i^*$ and $\mu = 0$. To see this, define $\hat{a}_i^v$ to be the level of voluntary abatement that solves the K-T condition (8a), which implies that $-C'\left(\hat{a}_i^v\right) / x'\left(\hat{a}_i^v\right) = \tau\varphi \sum_{k=1}^{K} \delta^k$. The cost to firm *i* of choosing $\hat{a}_i^v$ and then facing the tax policy is $C\left(\hat{a}_i^v, \theta_i\right) + \sum_{k=1}^{K} \delta^k \left[C\left(a_i^*, \theta_i\right) + \tau\varphi\left(x(\hat{a}_i^v, \theta_i) - x(a_i^*, \theta_i)\right)\right]$. We showed earlier that the cost of $a_i^v = a_i^*$ is $\sum_{k=0}^{K} \delta^k C\left(a_i^*, \theta_i\right)$. Thus the cost of $a_i^v = \hat{a}_i^v < a_i^*$ is lower than the cost of $a_i^v = a_i^*$ if $C\left(\hat{a}_i^v, \theta_i\right) + \sum_{k=1}^{K} \delta^k \tau\varphi\left(x(\hat{a}_i^v, \theta_i) - x(a_i^*, \theta_i)\right) < C\left(a_i^*, \theta_i\right)$. Substituting $-C'\left(\hat{a}_i^v\right) / x'\left(\hat{a}_i^v\right) = \tau\varphi \sum_{k=1}^{K} \delta^k$, rearranging terms and dividing each side by $a_i^* - \hat{a}_i^v$, the inequality becomes $\left[\frac{\left(x\left(a_i^*, \theta_i\right) - x\left(\hat{a}_i^v, \theta_i\right)\right)}{a_i^* - \hat{a}_i^v}\right] \Big/ x'\left(\hat{a}_i^v\right) < \left[\frac{\left(C\left(a_i^*, \theta_i\right) - C\left(\hat{a}_i^v, \theta_i\right)\right)}{a_i^* - \hat{a}_i^v}\right] \Big/ C'\left(\hat{a}_i^v\right)$, which must hold because of the assumed curvature of the cost and pollution functions. Therefore $a_i^v = a_i^*$ is not a best response for firm *i* when $\varphi < \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$. Thus, $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$ is necessary to induce optimal compliance in the voluntary stage game.

We have now shown that $\boldsymbol{a}^v = \boldsymbol{a}^*$ is part of a SPNE only when $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$. Next, we show that when $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$ then $\boldsymbol{a}^v = \boldsymbol{a}^*$ is the only possible NE in the voluntary stage. To see this, suppose that $\boldsymbol{a}_{-i}^v \leq \boldsymbol{a}_{-i}^*$, such that at least one firm is abating less than the socially optimal amount and firm *i* must overabate in order to meet the standard. Let ε > 0 denote the amount of overabatement needed by firm i to ensure that the standard is met so that $a_i^v = a_i^* + \varepsilon$ with an associated cost to firm i over K+1 periods of $\sum_{k=0}^{K} \delta^k C\left(a_i^* + \varepsilon, \theta_i\right)$. Firm i would never choose $a_i^v > a_i^* + \varepsilon$ since this would imply higher abatement costs without a reduction in tax burden. If firm i chooses $a_i^v < a_i^* + \varepsilon$ then the tax policy will be imposed. The optimal choice of voluntary abatement given that the tax policy will be imposed is determined by the cost minimization problem

$$\underset{a_i^v}{Min} \;\; C\left(a_i^v, \theta_i\right) + \sum_{k=1}^{K} \delta^k \left[C\left(a_i^*, \theta_i\right) + \tau\varphi\left(x(a_i^v, \theta_i) - x(a_i^* + \varepsilon, \theta_i)\right)\right] \;\; s.t. \;\; a_i^v \leq a_i^* + \varepsilon \, . \tag{9}$$

With corresponding K-T conditions

$$C'\left(a_i^v\right) + \tau\varphi \sum_{k=1}^{K} \delta^k x'\left(a_i^v\right) + \mu = 0 \tag{9a}$$

$$\mu\left(a_i^* + \varepsilon - a_i^v\right) = 0 \tag{9b}$$

$$\mu \geq 0 \, . \tag{9c}$$

We must consider solutions with either $\mu > 0$ and $a_i^v = a_i^* + \varepsilon$ or $\mu = 0$ and $a_i^v \leq a_i^* + \varepsilon$. Substituting $\tau = -C'\left(a_i^*\right) / x'\left(a_i^*\right)$ condition (9a) implies that $C'\left(a_i^v\right) + \mu = \varphi \sum_{k=1}^{K} \delta^k C'\left(a_i^*\right) \cdot \dfrac{x'\left(a_i^v\right)}{x'\left(a_i^*\right)}$. Further, since $\dfrac{x'\left(a_i^v\right)}{x'\left(a_i^*\right)} \geq 1$ we know that $C'\left(a_i^v\right) + \mu \geq \varphi \sum_{k=1}^{K} \delta^k C'\left(a_i^*\right)$. Therefore when $\varphi = \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$ then the only possible solution has $\mu = 0$ and $\hat{a}_i^v = a_i^*$, and when $\varphi > \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$ then solutions with $\mu > 0$ and $a_i^v = a_i^* + \varepsilon$ or $\mu = 0$ and $a_i^v \leq a_i^* + \varepsilon$ will always be possible.

Since $a_i^v < a_i^*$ will never be optimal, we restrict our focus to comparing the cost of choosing the optimal $a_i^* \leq \hat{a}_i^v < a_i^* + \varepsilon$ given that the tax will be imposed, to the cost of choosing $a_i^v = a_i^* + \varepsilon$ and thus avoiding the tax. The cost over K+1 rounds, associated with choosing $\hat{a}_i^v < a_i^* + \varepsilon$ is $C\left(\hat{a}_i^v, \theta_i\right) + \sum_{k=1}^{K} \delta^k \left[C\left(a_i^*, \theta_i\right) + \tau\varphi\left(x(\hat{a}_i^v, \theta_i) - x(a_i^* + \varepsilon, \theta_i)\right)\right]$ while the cost of choosing $a_i^v = a_i^* + \varepsilon$ and avoiding the tax policy is $\sum_{k=0}^{K} \delta^k C\left(a_i^* + \varepsilon, \theta_i\right)$. Substituting

$\tau \sum_{k=1}^{K} \delta^k \varphi = -C'\left(\hat{a}_i^v\right)/\, x'\left(\hat{a}_i^v\right)$ from the K-T conditions and rearranging, we have that the cost of choosing $\hat{a}_i^v$ is less than the cost of avoiding the tax if

$$-C'\left(\hat{a}_i^v\right)/\, x'\left(\hat{a}_i^v\right) < \frac{C\left(a_i^* + \varepsilon\right) - C\left(a_i^*\right)}{x\left(\hat{a}_i^v\right) - x\left(a_i^* + \varepsilon\right)} + \sum_{k=1}^{K} \delta^k \frac{\left[C\left(a_i^* + \varepsilon\right) - C\left(\hat{a}_i^v\right)\right]}{x\left(\hat{a}_i^v\right) - x\left(a_i^* + \varepsilon\right)}. \tag{10}$$

Multiplying the right hand side of equation (10) by $(\varepsilon/\varepsilon)$, the curvature of the cost and pollution functions imply that $C'\left(a_i^*\right) < \frac{C\left(a_i^* + \varepsilon\right) - C\left(a_i^*\right)}{\varepsilon} < \frac{C\left(a_i^* + \varepsilon\right) - C\left(\hat{a}_i^v\right)}{\varepsilon}$ and $-x'\left(a_i^*\right) \geq \frac{x\left(\hat{a}_i^v\right) - x\left(a_i^* + \varepsilon\right)}{\varepsilon}$. Since $\sum_{k=1}^{K} \delta^k > 0$ then equation (10) must hold and choosing $a_i^* \leq \hat{a}_i^v < a_i^* + \varepsilon$ is the minimum cost response for all values of ε. Therefore given that $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$, it is never a best response for a firm to overabate so that the standard is achieved. Given that other firms will not abate sufficiently to achieve the standard, it is therefore never in the best interest of any firm to underabate. The cost of abatement below a_i^* is lower than the cost of the tax, by definition, which implies that a firm that is currently underabating would always prefer increasing abatement rather than facing the tax. When $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$ we can never have an equilibrium in which one firm chooses $a_i^v < a_i^*$ and another firm chooses $a_i^v > a_i^*$ and therefore $\boldsymbol{a}^v = \boldsymbol{a}^*$ is part of a unique SPNE when $\varphi \geq \left(\sum_{k=1}^{K} \delta^k\right)^{-1}$. ■

Proposition 2: *If* $\tau = \lambda^*$ *then* $\{\boldsymbol{a}_i^v, \boldsymbol{a}_i^t\} = \{\boldsymbol{0}, \boldsymbol{a}_i^*\}$ *is never a SPNE when* $\varphi > 0$.

Proof of Proposition 2: Suppose that $a_{-i}^v = 0$. Firm *i* would then make its abatement decision based on the minimization problem in equation (9). Condition (9a), implies that $C'\left(a_i^v\right) + \tau\varphi \sum_{k=1}^{K} \delta^k x'\left(a_i^v\right) + \mu = 0$. Since $\tau > 0$ and $x'\left(a_i^v\right) < 0$ it follows that either $\mu > 0$, $C'\left(a_i^v\right) > 0$ or both. When $\mu > 0$ this requires that $a_i^v = a_i^* + \varepsilon > 0$ and when $C'\left(a_i^v\right) > 0$ this requires that $a_i^v > 0$. Therefore when all other firms choose zero abatement, firm *i* will optimally choose a positive level of abatement so that we can never have a zero abatement equilibrium. ■

III. Experimental Design

To test the relative performance of the voluntary/threat policies, a series of economics experiments were conducted at the Cornell Lab for Experimental Economics and Decision Research in the spring semester of 2006. Participants had taken at least one class in economics and the majority had participated in at least one prior (but unrelated) economics experiment. Experiment instructions were presented in writing, and orally with aid of PowerPoint slides. The experimental sessions lasted approximately one hour and participants earned experimental tokens during each decision round, which were exchanged for dollars at the end of the experiment at the announced rate of 70,000 tokens per $1US. Overall, there were 144 participants and average participant earnings were $20.

The experiment hierarchy is illustrated in Figure 1. There were six experimental treatments and each treatment was comprised of four separate experimental groups made up of six participants. The participants made decisions analogous to abatement decisions over 23 rounds[8] and the rounds were split up into Part A (rounds 1-5) and Part B (rounds 6-23). Part A was intended to establish a regulation-free baseline. Part B represented regulation under a voluntary-threat policy, whereby subjects faced one of the six policies listed in Table 1.

In each treatment, all participants faced the identical abatement cost function $C(a_i, \theta_i) = \delta a_i^{\alpha}$. As the term abatement implies reducing emissions relative to some benchmark, we instead framed the participants' decision as one of choosing a level of emissions. Specifically, emissions were related to abatement through the function $y_i = \gamma - a_i$. In addition, the abatement was related to ambient pollution through the linear function $x_i = \sum_{i=1}^{n}(\gamma - a_i)$.

Each participant was given an *"Emissions Decision Sheet"* that listed the "firm earnings" associated with all possible levels of emissions. To give policy relevance to the experimental parameters, the baseline ($a_i = 0$) firm earnings were chosen to proximate the net farm income of

[8] The actual number of rounds was random, however each group completed at least 23 rounds.

a medium sized dairy farm in New York State, operating with a herd size of 200 cows[9]. Table 2 lists the specific values for the experiment parameters, which conforms to the underlying assumptions of the theoretical model.

In Part B, each of the policy instruments were designed to induce a 40% reduction in ambient pollution levels, from an unconstrained profit-maximizing pollution level of 120 to an ambient standard, x^s, of 72. The 40% level was chosen so as to mirror the 40% nutrient reduction goals called for in the original Chesapeake Bay Agreement (CBP 2005). Reaching the ambient standard of 72 at least cost required each of the six participants to reduce their emissions to 12 units, from the unconstrained optimum of 20. This implies an optimal abatement amount, a_i^*, of 8 for each participant.

Under the tax policy, each participant pays a marginal tax, τ, of 2,500 tokens for every unit of ambient pollution, x^t, above the tax threshold, $\bar{x}$. Given that the marginal cost of reducing emissions beyond 12 is greater than 2,500 tokens and the marginal cost of emissions reductions by 12 units or less is less than 2,500, optimal abatement for each firm is exactly 12 units such that the ambient standard of 72 is exactly met.

In Treatment 1, the tax threshold is set equal to the ambient standard of 72. This duplicates the policy shown to be highly efficient in the experimental studies of Spraggon (2002), Poe et al. (2004), Cochard et al. (2004) and Suter et al. (2006) and serves as the baseline for evaluating the results of the voluntary/threat policy.

When τ =2,500, emitting exactly 12 units is a unique NE for any tax threshold at or below 72. However, when the tax threshold is strictly lower than 72 the *group* can maximize its payoff when participants emit fewer than 12 units. While collusive outcomes are not seen in recent experimental results when ambient pollution is a stochastic function of firm emissions (Suter et al. 2006), it is an open question whether participants behave in a more collusive manor in the non-stochastic environment presented in this study. Evidence from the closely related nonpoint pollution experiments of Spraggon (2002) suggests that, at least on average, decisions do not pivot on the presence/absence of uncertainty. In Treatment 2, the tax threshold is 50. which allows us to compare the results of the pure tax policy with the threatened tax policy of Treatment 4, which also has a tax threshold of 50.

[9] The average herd size and farm income amounts were determined based on the *New York State Dairy Farm Summary* reports produced by Cornell University for the years 1999-2003.

In Treatments 3 through 6 we evaluate the voluntary/threat policy with an ambient standard equal to 72 units. In each of these treatments, the threatened regulatory regime consists of three rounds (K=3). Three rounds were selected to allow for multiple observations of the voluntary scenario while still capturing the essence of a threat where participants pay a penalty over time for not meeting the standard, as suggested by Segerson and Wu. Given the short time frame over which the decision rounds occur, we assume that the discount factor, δ, is equal to 1.

In Treatments 3 and 4, the threatened regulatory policy has tax thresholds of 66 and 50 respectively. The threshold of 66 is low enough to provide the necessary incentives theoretically for voluntarily compliance. The threshold of 50 provides a stronger incentive for voluntary abatement, since the costs of the tax stage were higher, but also introduced the potential for a wider range of possible equilibria. By varying the tax threshold we gained some insight into the tradeoff between a tax threshold that is relatively close to the ambient standard and a lower tax threshold, which increases the incentive to abate voluntarily but also increases the potential for meeting the voluntary standard at higher than minimum cost.

Meeting the ambient standard voluntarily at least cost requires a great deal of coordination, since all participants must choose to emit exactly 12 units. In Treatment 5, we increase the potential for coordination by allowing groups to engage in costless, nonbinding communication (referred to in the experimental economics literature as "cheap talk"). In particular, each group is allowed up to five minutes of cheap talk before rounds 6, 11, 16 and 21. Participants are allowed to discuss any aspect of the experiment, but are not allowed to make threats or arrange for side payments. Cheap talk has been shown to greatly improve efficiency outcomes in earlier studies of the pure tax instrument (Suter et al. 2006).

In Treatment 6, we test the voluntary/threat policy with the endogenous threshold, whereby the zero abatement NE is eliminated. Recall from Section II that the choice of the scale parameter, φ, is in effect a choice of the magnitude of the incentive for voluntary compliance. To help engender transparency, we chose $\varphi = 1$, such that every unit of pollution above 72 in the voluntary stage results in the tax threshold being set an equal number of units below 72 in the tax stage.

IIIa. Testable Hypotheses

Although the pure tax, voluntary/threat with exogenous threshold, and voluntary/threat with endogenous threshold policies all theoretically induce outcomes whereby the ambient pollution standard is met at least cost, the relative empirical performance of the three mechanisms is an open question. While the pure tax policy with a constant marginal ambient tax has proven to generate highly efficient outcomes in several past experimental studies (Poe et. al. 2004, Spraggon 2002, Cochard et. al. 2004; Suter et. al. 2006), a voluntary policy with a threat of regulation has not been participant to experimental examination. By evaluating the experimental results from the voluntary/threat policy we endeavor to test the following three hypotheses.

(1) In the voluntary/threat policy treatments, firms abate voluntarily such that the ambient pollution standard is met.

(2) Firms are more likely to abate voluntarily with a lower threatened tax threshold, $\bar{x}$, and when the threshold is endogenous.

(3) The instances of participants choosing zero abatement are lower in the voluntary policy with an endogenous as opposed to an exogenous threshold.

Comparing the results from the pure tax treatments to the results in the voluntary/threat treatments, we then test two additional hypotheses:

(4) The average emissions decision in each of the policy settings is identical to the NE predictions.

(5) The voluntary/threat policies generate social efficiency outcomes identical to the outcomes under the tax only policy.[10]

IV. Results

In this section we present three sets of results. We begin with a simple presentation of the outcomes from the four voluntary/threat treatments. This presentation includes the number of rounds that each group met the ambient pollution standard voluntarily as well as evidence on how individual behavior differs across treatments. Based on these results, we draw conclusions regarding the first three hypotheses above. The second set of results relies on an econometric model to estimate the mean participant-level emissions decision in each of the policy scenario. This enables us to draw conclusions regarding Hypothesis 4. In the final set of results, we present

[10] The notion of efficiency is odd here given that we are in a cost-effectiveness framework. However, the efficiency calculations do allow for more delicate comparisons both across treatments and with related studies.

social efficiency outcomes for the six treatments. The efficiency results allow for a general comparison across all six treatments and specifically allow us to compare the outcomes of the tax only treatments relative to the voluntary/threat treatments.

For all of the results presented below, our analysis covers the decisions made up to round 23. This implies that for each participant we have 5 observations from Part A and 18 Part B observations. In addition to the summary results presented below, we also include a round by round graphical depiction of the group emissions for all treatments as an Appendix.

Result 1: In the absence of communication, participants generally do not meet the ambient standard voluntarily. With communication, we fail to reject the null hypothesis that groups comply voluntarily.

Inspection of Table 3 reveals that when groups are not allowed to communicate they have a very difficult time meeting the ambient pollution standard in the voluntary stage. When groups are allowed to communicate in Treatment 6, however, they are able to reach the ambient standard voluntarily with great regularity [11].

Result 2: Decreasing the threshold in the exogenous threat mechanism setting and implementing an endogenous threshold increase the likelihood the ambient standard is met voluntarily. Further, there is no strong evidence of free-riding.

While lowering the tax threshold from 66 to 50 increases the probability of voluntary compliance, groups still violate the standard more often than not. Even making the threshold endogenous, a case where voluntary compliance is the unique SPNE, only one of the four groups meet the standard in a voluntary round. Interestingly, the endogenous threat policy is the only scenario where a group exceeded the standard in the first voluntary round, but then met the standard voluntarily after experiencing the tax policy.

Measuring the degree of free riding when the standard was met is challenging, primarily because groups generally do not achieve the standard voluntarily. From the limited evidence available, it appears that free riding is not an issue. We expect the greatest potential for free

[11] Although group 2 did fail to meet the standard voluntarily in one of the rounds (and subsequently had to go through three rounds of the regulatory policy), this was a result of a mistake made by one of the participants. In the cheap-talk session that occurred after the mistake was made, the participant was apologetic to the other group members and stated that the wrong number was accidentally typed into the computer.

riding to occur with the voluntary policy and the exogenous threat mechanism with a threshold of 50, however we observe only one round where one participant overabated and one participant underabated (out of 23 rounds where the standard was met voluntarily). In the group that met the standard under the voluntary policy with the endogenous threat, we observe one participant that was consistently one or two units below the optimal emissions and one consistently one or two units above. This limited evidence does not indicate the prevalence of the more drastic types of free riding that are theoretically possible.

Result 3: The endogenous threshold mechanism does reduce the incidence of choosing the zero abatement strategy. Lowering the tax threshold of the exogenous threat mechanism also reduces the frequency at which firms choose zero abatement.

To arrive at Result 3 we measure the frequency at which the zero abatement strategy is chosen (i.e. when a participant chooses to emit 20 or more units). In the voluntary treatment with the exogenous threshold of 66, participants play the zero abatement strategy an average of 15 times per group over the 18 observed Part B rounds (s.e.=4.5). The per group frequency dropped to 5.3 (s.e.=2.3) in the case of the threatened exogenous threshold of 50. Finally, in the voluntary with endogenous threshold treatments the average number of times the zero abatement strategy was played dips to 4.5 times per group (s.e.=2.3). Note that only when the threshold is endogenous is the average number of participants in a group that violates the voluntary standard not significantly different from zero.

In addition to looking at the number of instances where participants chose not to abate in the voluntary rounds of the experiment, it is also interesting to look at the decisions participants made in the first round in which the voluntary/threat policy is in effect (round 6). In this round each participant has to make a decision without any prior information on how other participants would respond to the policy and therefore we get a test of the initial effectiveness of the voluntary/threat instrument.

Interestingly, the majority of participants voluntarily abated at least the optimal amount. For each of the three treatments there were 24 observed decisions in round 6. The number of participants that chose to emit 12 or fewer units was 17 (tax threshold 66), 21 (tax threshold 50) and 20 (endogenous tax threshold). The fact that over 80% of participants voluntarily abated at or above the amount necessary to reach the standard is a testament to the fact that the threat of the regulatory policy was strong.

Two intuitive hypotheses could explain why the remaining participants refused to adequately abate. First, it is possible that they believed that other participants would not abate and therefore it was not in their best interest to abate. Second, they were either confused or they made a miscalculation regarding the payoffs of the various strategies. While we cannot make a definitive statement on this matter, it appears that the latter explanation is most accurate. Evidence in support of this conclusion comes from the endogenous threat treatment, where participants should abate even if they believe that others will not follow suit. Further, we have some limited evidence from an experiment we ran where the ambient standard was based on individual rather than group emissions. In this case, it is always optimal to emit 12 units, since whether or not you meet the ambient standard is only a function of your own decision. In this sub-treatment we observed 2 of the 8 participants not abate in the first policy round, approximating the twenty percent of participants that did not abate in the group setting.

Having twenty percent of participants make a miscalculation in the initial period of a policy does not seem as if it should be an overwhelming obstacle. In many experimental settings it takes numerous decision periods and substantial learning before theoretically optimal outcomes are achieved (if they are achieved at all). In that regard, it is important to investigate the evolution of decisions over time. In addition, since we have seen from Table 3 that the majority of groups did not meet the standard voluntarily, it is important to examine what happens in the tax policy rounds of the experiment. We can then compare the voluntary/threat mechanism results under both the voluntary and the tax policy settings, to the pure tax settings of Treatments 1 and 2.

To get a sense of how individual decisions varied across treatment conditions, our objective is to generate an expectation for the emissions decision of a random participant in a random group in one of the treatment scenarios. Recall from the last section that the individual observations come from a hierarchical data generating structure, where groups were nested within treatments, participants were nested within groups and each participant made a decision over a series of rounds. Additionally, in the voluntary/threat treatments, participants either made a decision in a voluntary setting or in a tax policy setting. This complex data structure implies that we cannot treat each of the individual decisions as an independent observation. It is reasonable to presume that there is serial correlation among the individual decisions across rounds and that the individual decisions within a round are correlated across the six participants in a group.

To address the complications having to do with the fact that groups that meet the ambient standard voluntarily will necessarily participate in more voluntary rounds, we calculated the mean emissions decision made by each subject in Part A and the mean emissions decisions made in the voluntary and the tax policies of Part B for three aggregate round groupings. The three aggregate groupings correspond to rounds 6-11, 12-17 and 18-23. By aggregating, we ensure that the voluntary decisions made by each individual are weighted equally.

To compare the individual emissions decisions across treatments we then estimate three mixed models. In each of the models, the general structure can be written as

$$y_{igr} = X\beta + \varepsilon_{igr} \ . \tag{11}$$

where y_{igr} is the emissions decision made by individual i in group g and treatment r. In the above formulation, X is known as the design matrix, which is a matrix of 1's and 0's used to represent the form of the fixed effects, and β is a vector of fixed effect coefficients to be estimated. The model error is represented by ε_{igr}.

In the first mixed model we use the data from the decisions made in Part A and specify a fixed effect for each of the six treatments[12]. The error term in the first mixed model is assumed to be independent, homogeneous and follow a normal distribution.

The second mixed model that we estimate uses data from the tax policy rounds of Part B. Here we include fixed effects for the three aggregate round groupings, the individual treatments and the interaction between the round groupings and the treatments. Additionally, we include a random effect α_g to account for the fact that individuals make decisions within groups. We assume the model error, ε_{igr}, to be serially correlated across the three round groupings and assume that this correlation follows an AR(1) process. Finally, since the variability of the emissions decisions are likely not equal across treatments, we allow the error variance and the correlation coefficient, ρ, to be treatment specific. Utilizing the identical structure, the third mixed model is estimated using the emissions decisions made under the voluntary policy. As such, the tax only treatments are not included in this model.

[12] There is insufficient variation at the group level to include a random group effect.

The coefficients in equation (11) for each of the threes models are estimated using the SAS proc mixed command and included in Table 4. The results show that in Part A of the experiment, when no policy was in place, emissions were not different from the prediction of 20. In other words, with no regulatory policy or threat of a regulatory policy in place, participants on average reached the unconstrained optimum. In each of the Part B policy scenarios individual emissions were significantly below 20, which suggest that all of the policies that we investigated resulted in positive levels of abatement. Individual emissions are, however, significantly different from the NE prediction, which is also the socially optimal emissions decision, of 12 in at least one aggregate round grouping of each of the policy scenarios.

Result 4: In all of the policy scenarios without communication, mean decisions deviate from the Nash prediction in at least one period.

In the pure tax setting with the threshold set equal to the ambient standard, which serves as our baseline, the estimated emissions decision is significantly higher than the NE of 12. Despite the statistical significance of this outcome, the fact that participants on average exceed the optimum by 10% is not large in economic terms and closely approximates earlier experiments by Spraggon (2002) and Poe et. al. (2004)[13].

When the tax threshold equals 50, the expected decision in the pure tax policy is significantly less than the NE in all three round groupings, indicating a tendency towards tax avoidance through excess abatement. The group's after tax profits are maximized when each participant emitted 8 units, however without communication we expect that individuals attempting to maximize individual profits would drive the average results towards the NE result of 12 units of emissions. Though limited evidence indicative of tacit collusive does occur, it erodes between the earlier and later rounds (although the erosion was not statistically significant).

In the voluntary/threat policy with an exogenous threshold of 66, the mean individual emissions level of 14.32 in the early rounds of the voluntary setting is significantly greater than the NE. Additionally, individual emissions increase significantly between early and late rounds, indicating that the response to the tax threat grew weaker over time as participants became

[13] Spraggon calculates average individual emissions under the pure tax mechanism to be 31.84 when the social optimum is 25. Poe et. al. calculate an individual average of 5.93 when the social optimum is 5.

convinced that other group members would also not abate voluntarily. In the tax stage of the treatment, emission levels are not significantly different from the NE in either early or late rounds.

In the voluntary/threat policy with an exogenous threshold of 50, individual emissions average 12.67 and are not significantly different from 12 in the early voluntary rounds, however we do again see a significant increase in later rounds, as expected emissions increase to 14.21. Mean emissions under the tax policy range from 9.26 to 10.14 and are significantly less than 12, similar to our results in the pure tax policy, which again suggests some tendency towards tax avoidance that may serve to weaken the tax threat.

When the threatened tax threshold is endogenous, estimated emissions in the early rounds average12.53 under the voluntary policy and 12.05 under the tax policy. These figures are closer to the social optimum of 12 than with any of the other treatments without communication. Further, the change in expected emissions is not significant between early and late rounds in either the voluntary or tax setting, although estimated emissions are significantly different from 12 and in the late round groupings of the voluntary policy.

Comparing the voluntary/threat policy to the pure tax policy we see that in the voluntary stage the expected emissions when the threatened tax threshold is set at 50 or is endogenous are not different from the baseline tax policy. In the tax stage of the voluntary/threat policy, expected emissions are statistically less than the baseline tax policy. As mentioned earlier, the threshold of 66 and the endogenous threshold yield expected emissions that are not significantly different from the NE. The endogenous threat therefore mechanism performs as well or better than the baseline tax policy in both the voluntary and tax portions of the policy.

Estimating individual emissions decisions is important in understanding how the various policy scenarios influence ambient pollution, however, they do not tell us how the voluntary/threat policies compare to the pure tax policy from the perspective of social efficiency. For example, if a policy approximates the cost minimizing emissions level for an average participant, this does not mean that costs have been minimized if there is a significant degree of variation around the mean decision.

To facilitate a more intricate comparison of decisions across treatments, we compute efficiency measures for each treatment assuming a damage function that is linear in ambient pollution, with a slope of 2,500. The choice of damage function does not have a significant impact on the relative efficiencies across the treatments and the choice of a linear damage function is consistent with previous experimental analyses (e.g. Spraggon 2002; Poe et. al. 2004).

The efficiency measure is identical to that of Spraggon (2002). The economic surplus in a given round is determined by summing the pre-tax earnings of each of the six firms (the social benefit) less the social damage, determined by the ambient pollution in that round. The observed surplus in round *t* by group g, S_{gt}, is then measured against the surplus in the zero abatement scenario, S_{zero}, and the maximum surplus possible, S_{max}, to give a measure of efficiency according to the formula

$$Efficiency_{gt} = \frac{S_{gt} - S_{zero}}{S_{max} - S_{zero}}. \tag{12}$$

The mean efficiency measures for each treatment in each of the three aggregate round groupings are then compared using the mixed modeling procedure described for the individual emissions, except that the level of observation is at the group rather than the individual level. We again assume that the model error term follows an AR(1) process and allow the correlation coefficient as well as the error variance to vary across treatments. The estimated mean efficiency levels and standard errors are given for each treatment in Table 5 in addition to the estimated error variances and correlation coefficients.

Result 5: The social efficiency outcomes for the voluntary/threat treatments are either not statistically different or are higher than the outcomes in the pure tax treatments.

The three voluntary/threat treatments where participants without communication all have mean efficiencies that are not significantly different from the tax only baseline, though the voluntary policy with endogenous threat has the highest efficiency of the three. With, the efficiency results are significantly higher than the baseline (and are not significantly different from 100%). In the tax only setting with the threshold equal to 50, the mean efficiencies were significantly lower than all of the other treatments.

V. Conclusion

In this paper we evaluate a policy introduced Segerson and Wu (2006) that addresses nonpoint source pollution through a voluntary policy used in combination with a background threat of regulation. In addition, we augment Segerson and Wu's theory by showing how the severity of the threatened regulatory policy can be made endogenous by conditioning on decisions made in the voluntary period, which removes the existence of an equilibrium whereby all firms abate nothing in the voluntary period. Using results obtained in the experimental economics laboratory, we show that although lowering the threatened exogenous tax threshold or making the threshold endogenous increases the probability that a group of polluting firms meet the ambient standard voluntarily, the standard is still only met by approximately 25% of the groups in the absence of communication. When participants are allowed to communicate the probability that a group meets the ambient standard voluntarily improves to nearly 100% with essentially no free-riding. This positive result supports the findings of Poe et al. (2004) and Vossler et al. (2006) that communication can greatly improve the social efficiency of ambient tax mechanisms.

Despite the fact that the majority of groups do not meet the ambient standard voluntarily in the absence of communication, the political attractiveness of allowing firms the opportunity to meet a standard voluntarily remains. In addition, in the approximately 25% of groups where the ambient standard is met voluntarily, policy makers would not need to expend the resources necessary to determine a theoretically optimal marginal tax.

When the ambient standard is not met voluntarily, the tax system that is imposed still results in the ambient standard being theoretically met at least cost. Our experimental results suggest, however, that this may not necessarily be the case. Lower levels of the tax threshold appear to introduce a greater potential for over abatement and increased variance in the tax setting and may therefore be undesirable. The relatively high exogenous tax threshold does not provide a strong enough incentive to convince individuals to abate voluntarily while the relatively lower threshold induces a greater number of subjects to abate voluntarily, but also exhibits over abatement under the tax policy in the groups that do not meet the ambient standard voluntarily. When the threshold in the tax policy is made endogenous, however, groups are more likely to engage in voluntary compliance the groups that are non-compliant exhibit near-optimal abatement levels in the tax policy.

Finally, our results show that in the three voluntary/threat policy treatments without communication, measured social efficiency levels ae not significantly different from the baseline pure tax policy advocated by previous studies. The voluntary/threat policy provides significant political advantages in that landowners have an opportunity to reduce emissions voluntarily and avoid direct regulation. Therefore the voluntary/threat policy may be both more politically palatable and at the same time generate economic outcomes that are as good as or better than a strictly tax based mechanism.

The optimistic viewpoint that we take regarding the voluntary/threat policy certainly coul be bolstered with future research on variations of the policies tested here. Specifically, it would be interesting to see if increasing the severity of the endogenously determined threat by varying the scale could improve the probability that groups meet the standard voluntarily. Further, we did not examine how changing the number of rounds (*K*) or adding a stochastic component to the experiment influences the probability that an individual will abate optimally in both the voluntary and tax settings.

References

Alpízar, Francisco, Till Requate, and Albert Schram. 2004. Collective versus Random Fining: An Experimental Study on Controlling Ambient Pollution. *Environmental and Resource Economics* **29**: 231-252.

Cabe, Richard, Joseph A. Herriges. 1992 The Regulation of Non-Point Sources of Pollution Under Imperfect and Asymmetric Information. . *Journal of Environmental Economics and Management* **32**: 134-146.

Chesapeake Bay Program. Updated 2005. Overview of the Bay Program. Available: http://www.chesapeakebay.net/overview.htm.

Cochard, Willinger, Xepapadeas**.** 2005. Efficiency of Nonpoint Source Pollution Instruments: An Experimental Study. 30: 393-422 *Environmental and Resource Economics* **29**: 231-252.

Dairy Farm Management: Business Summary New York State. 1998-2002. Department of Applied Economics and Management. Cornell University. Ithaca, NY.

Economic Research Service. Updated 2002. U. S. Department of Agriculture. ERS Analysis: Conservation Programs. Available: http://www.ers.usda.gov/Features/FarmBill/analysis/conservationoverview.htm

Hansen, Lars G. 1998. A Damage Based Tax Mechanism for Regulation of Non-Point Emissions. *Environmental and Resource Economics* **12**: 99–112.

Horan, Richard D., James S. Shortle, and David G. Abler. 1998. Ambient Taxes when Polluters have Multiple Choices. *Journal of Environmental Economics and Management* **36**: 186-199.

Karp, Larry. 2005. Nonpoint Source Pollution Taxes and Excessive Tax Burden. *Environmental and Resource Economics* **32**: 229–251.

National Resources Inventory. 2002. 2002 Annual NRI. United States Department of Agriculture. Natural Resources Conservation Service. Available: http://www.nrcs.usda.gov/technical/land/nri02/

Poe, Gregory L., William D. Schulze, Kathleen Segerson, Jordan F. Suter and Christian A. Vossler. 2004. Exploring the Performance of Ambient-Based Policy Instruments when Non-Point Source Polluters can Cooperate. *American Journal of Agricultural Economics* **86**: 1203-1210.

Ribaudo, Marc. 2003. Agricultural Resources and Environmental Indicators, 2003. Chapter 2.3 Water Quality Impacts of Agriculture. Agricultural Handbook AH722. United States Department of Agriculture. Economic Research Service.

Ribaudo, Marc. 2004. Policy Explorations and Implications for Nonpoint Source Pollution Control: Discussion. *American Journal of Agricultural Economics* **86**: 1220-1221.

Segerson, Kathleen. 1988. Uncertainty and Incentives for Non-Point Source Pollution. *Journal of Environmental Economics and Management* **15**: 87-98.

Segerson, Kathleen, Thomas J. Miceli. 1998 Voluntary Environmental Agreements: Good News for Environmental Quality? *Journal of Environmental Economics and Management* **36**: 109–130.

Segerson, Kathleen, JunJie Wu. 2006. Nonpoint Source Pollution Control: Introducing First Best Outcomes Through the Use of Threats. *Journal of Environmental Economics and Management* **51**: 165-184.

Shortle, J., D. Abler, and M. Ribaudo. 2001. Agriculture and Water Quality: The Issues. In J. Shortle and D. Abler (Eds.) *Environmental Policies for Agricultural Pollution*. Walingford, UK: CAB International.

Spraggon, John. 2002. Exogenous Standarding Instruments as a Solution to Group Moral Hazards. *Journal of Public Economics* **84**: 427-456.

Spraggon, John. 2004. Testing Ambient Pollution Instruments with Heterogeneous Agents. *Journal of Environmental Economics and Management* **48**: 837-856.

Suter, Jordan F., Christian A. Vossler, Gregory L. Poe, and Kathleen Segerson. 2006. Experiments on Damage-Based Ambient Taxes for Nonpoint Source Polluters. .Working Paper, Department of Applied Economics and Management, Cornell University.

U.S. Environmental Protection Agency. 2002. *National Water Quality Inventory 2000 Report*. Office of Water, Washington, DC.

U.S. Environmental Protection Agency. 2006. *National Streams Report.* Wadeable Streams Assessment. Available: http://www.epa.gov/owow/streamsurvey/

Vossler, Christian A., Gregory L. Poe, William D. Schulze, and Kathleen Segerson. Forthcoming. Communication and Incentive Mechanisms Based on Group Performance: An Experimental Study of Nonpoint Pollution Control*.* *Economic Inquiry* **44**: 599-613.

Xepapadeas, Anastasios. 1991. Environmental Policy under Imperfect Information: Incentives and Moral Hazard. *Journal of Environmental Economics and Management* **20**: 113-126.

Figures and Tables

Figure 1: Hierarchical structure of experiment

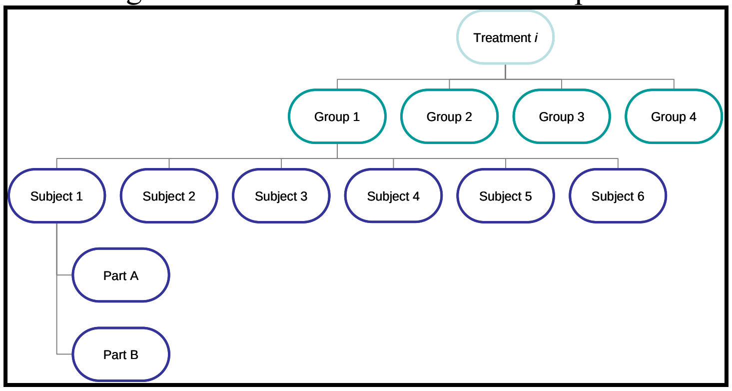


Table 1: Treatment Summary

	Policy Scenario	**Tax Threshold** $(\bar{x})$	**Communication Allowed**
Treatment 1	Tax Only	72	No
Treatment 2	Tax Only	50	No
Treatment 3	Voluntary/Threat	66	No
Treatment 4	Voluntary/Threat	50	No
Treatment 5	Voluntary/Threat	50	Yes
Treatment 6	Voluntary/Threat	Endogenous	No
4 groups per treatment, 6 participants per group, 138 total participants			

Table 2: Experimental Parameters

Description	**Functional Form**	**Parameter Values**
Abatement Cost Function	$\delta(\boldsymbol{a_i})^{\alpha}$	$\delta = 13 \quad \alpha = 3$
Firm Earnings	$\boldsymbol{Y} = \boldsymbol{Y}^0 - \delta(\boldsymbol{a_i})^{\alpha}$	$\boldsymbol{Y}^0 = 75{,}000$
Firm Level Emissions	$x_i = \gamma_i - a_i$	$\gamma_i = 20$
Ambient Pollution	$x = \sum_{i=1}^{n}(\gamma_i - a_i)$	$\boldsymbol{n} = 6$
Regulatory Only Policy	Tax payment $= max[\tau(x - \bar{x}), 0]$	$\tau = 2{,}500$ $\bar{x}$ = See Table 1
Voluntary/Threat Policy	<u>Voluntary Round</u> Tax payment = 0 <u>Regulatory Round</u> Tax payment $= max[\tau(x - \bar{x}), 0]$ (*instituted for* $\boldsymbol{K}$ *rounds if* $\boldsymbol{X} > \boldsymbol{X}^S$ *in voluntary round*)	$\tau = 2{,}500$ $\bar{x}$ = See Table 1 $\boldsymbol{K} = 3$

Table 3: Number of Part B rounds in which the ambient standard was met voluntarily

	No Communication			Communication
	$\bar{x}$ = 66	**$\bar{x}$ = 50**	**$\bar{x}$ = End.**	**$\bar{x}$ = 50**
Group 1	6	0	0	18
Group 2	0	5	0	14
Group 3	0	18	0	18
Group 4	0	0	14	18
Average	*1.50*	*5.75*	*3.50*	*17*
(standard deviation)	*(3.0)*	*(8.5)*	*(7.0)*	*(2.0)*

Table 4: Estimation Results

Dependent Variable: Mean Individual emissions
Number of Observations: 756

Tax Threshold	**Part A Rounds 1-5**	**Policy Scenario**	**Part B Rounds 6-11**	**Rounds 12-17**	**Rounds 18-23**	$\sigma^2_{T,V}$	ρ
72	20.02 *(0.337)*	**Tax Only**	13.82* *(0.526)*	13.07* *(0.526)*	13.72* *(0.526)*	5.85	0.766
50	19.76 *(0.337)*		9.70* *(0.527)*	10.31* *(0.527)*	10.75* *(0.527)*	6.91	0.718
66	19.24 *(0.337)*	**Voluntary**	14.32* *(0.986)*	16.29* *(0.986)*	17.42* *(0.986)*	11.88	0.565
		Tax	11.42 *(0.363)*	11.39 *(0.356)*	11.56 *(0.356)*	2.25	0.916
50	19.94 *(0.337)*	**Voluntary**	12.67 *(0.932)*	13.50* *(0.932)*	14.21* *(0.932)*	9.42	0.568
		Tax	9.44* *(0.654)*	9.26* *(0.654)*	10.14* *(0.654)*	8.61	0.859
Endogenous	19.69 *(0.337)*	**Voluntary**	12.53 *(0.881)*	13.15 *(0.881)*	13.69* *(0.881)*	7.18	0.675
		Tax	12.05 *(0.612)*	11.86 *(0.636)*	11.44 *(0.653)*	8.18	0.860
50 (Com)	19.73 *(0.337)*	**Voluntary**	12.00 -	12.00 -	12.10 -	-	-
		Tax	- -	- -	- -	-	-
		Estimated Group Level Variance				**Tax** = 0.132 **Voluntary** = 1.91	

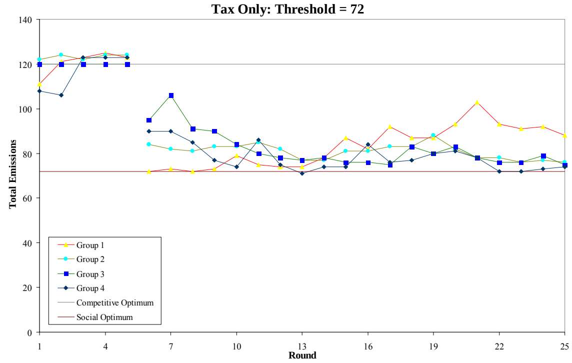
* Indicates that the coefficient estimate in Part B is significantly different from 12 at the 5% level. None of the results from Part A are significantly different from 20.

Table 5: Mean efficiency levels by treatment

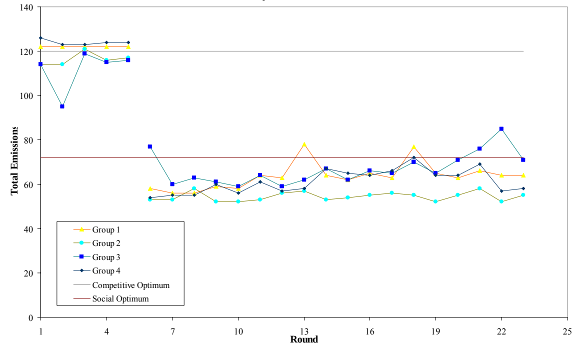
Tax Threshold	**Policy**	**Rounds 6-11**	**Rounds 12-17**	**Rounds 18-23**	$\sigma^2_{T,V}$	ρ
72	Tax Only	78.85 *(6.50)*	85.27 *(6.50)*	80.25 *(6.50)*	0.017	0.296
50	Tax Only	79.05 *(9.87)*	74.51 *(9.87)*	72.51 *(9.87)*	0.039	0.872
66	Voluntary/Threat	81.04 *(3.03)*	85.47 *(3.03)*	73.31 *(3.03)*	0.004	0.511
50	Voluntary/Threat	57.86 *(6.11)*	64.18 *(6.11)*	70.71 *(6.11)*	0.015	0.587
Endogenous	Voluntary/Threat	84.25 *(8.02)*	82.97 *(8.02)*	79.85 *(8.02)*	0.026	0.960
50 (Com)	Voluntary/Threat	100.00 *(2.99)*	100.00 *(2.99)*	94.82 *(2.99)*	0.004	-

Appendix

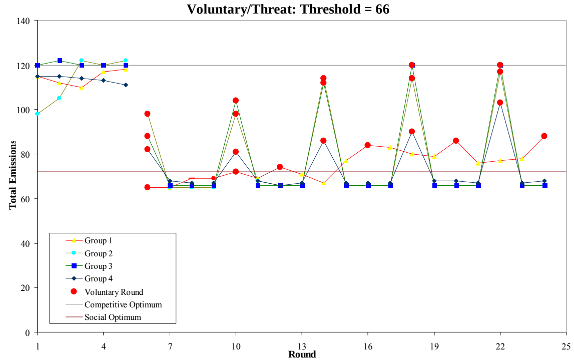
Total Pollution by Round
Tax Only: Threshold = 72



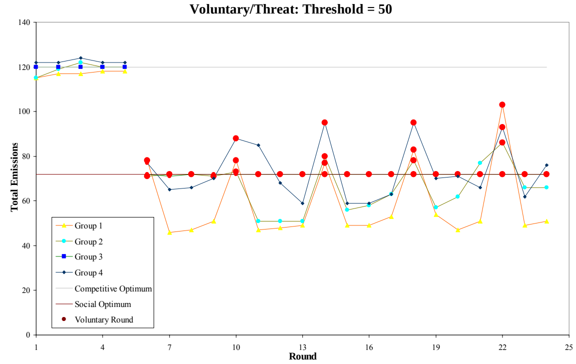
Total Emissions by Round
Tax Only: Threshold = 50



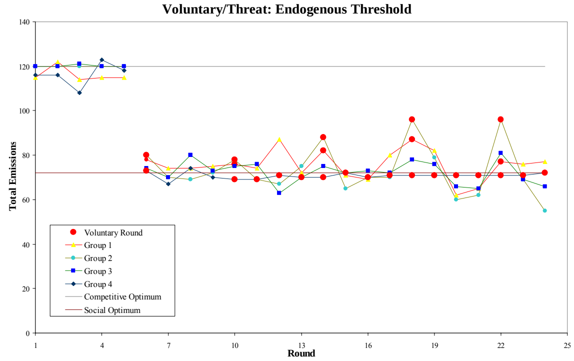
Total Emissions by Round
Voluntary/Threat: Threshold = 66



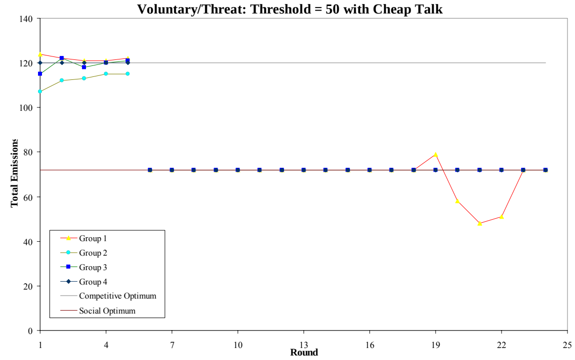
Total Emissions by Round
Voluntary/Threat: Threshold = 50



Total Emissions by Round
Voluntary/Threat: Endogenous Threshold



Total Emissions by Round
Voluntary/Threat: Threshold = 50 with Cheap Talk



Choice Experiments to Assess Farmers' Willingness to Participate in a Water Quality Trading Market

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Introduction

Water Quality Trading (WQT) has received increased attention as a means to achieve water quality goals. Several such trading programs have been adopted in several states throughout the nation, with more than 70 programs now in operation (Breetz et al., 2004). In principle, such programs could be applied to any water-borne pollutant and allow trading among point sources, among nonpoint sources, or between point and nonpoint sources (the latter is known as 'point-nonpoint trading'). Most of the existing programs are designed with point-nonpoint trading to limit nutrient loading: point sources are allowed to meet their nutrient emission limits by purchasing water quality credits from agricultural producers in the surrounding watershed. These producers are then obligated to implement a best management practice (BMP) that reduces expected nutrient loading by an amount commensurate with the number of credits sold.

Substantial evidence exists that nonpoint sources can reduce nutrient loading at a much lower cost than point source polluters in many watersheds. This suggests that a well functioning WQT program would be a more cost-effective strategy for meeting total maximum daily load requirements than regulating point source polluters alone (Faeth, 2000). The potential for pollution trading to lower control costs has already been realized in the active air quality trading markets (NCEE, 2001).

Despite the potential gains from WQT, perhaps the most commonly noted feature of existing programs is low trading volume; none of the programs have had extensive trading activity and many have had no trading at all (Hoag and Hughes-Popp, 1997). Our particular interest in this paper is the participation of nonpoint sources, almost always agricultural crop producers in existing programs. The reluctance of farmers to participate in WQT reflects a broader reluctance to adopt environmental practices in exchange for monetary payments (e.g., Cooper and Keim 1996).

Evidently, farmers perceive some intangible costs of participating in WQT markets that are not offset by the monetary gains from trading. These costs may include the disutility of the managerial effort required to maintain BMPs, and/or a distaste for the WQT market procedures and rules. For example, farmers may object to the intrusiveness of being inspected or monitored to ensure their BMP is in place, or find the process of signing up for the program to be too onerous.

Although the existence of intangible costs is apparent from empirical evidence, the factors giving rise to these costs are not well understood. The objective of this paper is to quantify the impact of different institutional factors on farmer's stated behavior in a WQT market. In particular, we wish to determine the importance – relative to monetary trading income – of various WQT market attributes on farmers' willingness to participate in such a market. The magnitude of these factors will provide information about how to design a program to encourage participation and, more broadly, will identify the situations where a WQT market is feasible given that certain rules are necessary.

The method of choice experiments is well suited to our research question. Choice experiments were originally developed in the marketing literature in order to determine the implicit market value of various product attributes. Subjects in these experiments make a choice from a side-by-side comparison of 3 or more products, which vary by different attributes including price. The choice data is then analyzed using discrete choice regression models, such as conditional logit, to estimate the effect of each attribute on the probability that the consumer chooses the product. This method has been widely adopted by environmental economists studying choice behavior related to environmental quality, such as selection of recreation sites (e.g., Adamowicz et al., 1997) and housing location (e.g., Earnhart, 2001). Economists studying agricultural markets have also applied the method to understand the attributes of food products influencing consumers' shopping choices (e.g., Fox et al., 2002).

This paper describes a set of choice experiments designed to elicit WQT trading behavior of Great Plains crop producers in different situations. In our case, the attributes to be varied across choices are the features of trading, such as the effort required for signup and the monitoring the farmer would need to undergo. Choice experiments are being conduced in person with producers at events in different locations in Kansas from August 2006 through January 2007.

Only our first set of choice experiments has been completed to date. After describing the design of our choice experiments and the data collection procedures, we present an initial analysis of the small dataset assembled so far. This analysis is based on only simple, descriptive methods and is intended primarily to validate our data collection procedures. In addition, we collected qualitative data (written responses to open-ended opinion question), which provide insight on the appropriate model specification for our future econometric analysis.

Experimental Design

The purpose of our experiments is to identify market rules and attributes that influence farmers' willingness to participate in a point-nonpoint WQT market. After reviewing the operations of existing programs and consulting with Extension personnel and a small group of farmers in Kansas, we identified four market attributes that are likely to affect participation: (1) application time and effort, (2) the monitoring method, (3) penalties for violations, and (4) the BMP to be adopted. Embedded within the definition of BMPs is another key attribute: the degree of flexibility a farmer would have in fulfilling his trading obligations. As noted above, the price of credits is an additional explicit attribute, which will ultimately allow us to compute the implicit values of the other four. These attributes are listed in Table 1 and are described in more detail below.

Table 1. Design Attributes and Levels

Attribute	Variable Name	Levels
Application Time (hours)	*Time*	4, 16, 24, 40
Monitoring method	*Monitoring*	Annual verification, Spot check
Penalty ($/acre enrolled)	*Penalty*	50, 100, 250, 500
Annual trading revenue ($/acre enrolled)	*Revenue*	3, 7, 15, 25
Best Management Practice	*BMP*	Filter strip (no haying/grazing), Filter strip (with haying/grazing), 100% No-till, Rotational No-till

By designing our experiments with different levels of our five attributes, we generate a dataset that allows us to test whether the institutional attributes affect trading choices, and if so, the magnitude of these impacts relative to price. Farmers were asked to choose among different opportunities to trade, which varied across the five attributes. Such choice scenarios would arise in an actual trading program, for example, if a WQT program were established in some region that allowed buyers to spell out the terms of the trading contract. Different buyers would then develop different contracts suiting their needs, giving rise to a range of trading opportunities for farmers. In the choice experiment method, the attributes are varied systematically based on experimental design principles, so that the resulting dataset maximizes statistical efficiency. In what follows, we describe the attributes we vary in our choice experiments and then explain the procedures we followed to design our choice sets.

Design Attributes

This section describes each of the attributes varied in our experiments and rationale for the levels we selected (Table 1). As noted above, trading opportunities are defined as different combinations of these attribute levels. A sample choice scenario presented to farmers is in Figure 1. Each scenario asks farmers to choose one of two trading opportunities, labeled Option A and Option B, or else choose Option C - "do not enroll." To facilitate comparison, all trading opportunities were assumed to be for a 10-year contract on a 100-acre field.

Scenario 8

You have two opportunities to sell credits in a Water Quality Trading market, given by Option A and Option B below. Your choices are to enroll your entire 100-acre field in one of these options (but not both) or neither of them.

	Option A	Option B	Option C
Application time (hours)	24	40	Do Not Enroll
Monitoring method	Annual verification	Annual verification	
Penalty for violations ($/acre enrolled)	100	100	
Best Management Practice (BMP)	Filter strip (with haying/grazing)	Rotational no-till	
Price and Cost information			
Offer price per credit ($/credit/year)	$2.50	$1.40	
Credits generated per acre enrolled	6	5	
Credit Revenue ($/acre/year)	$15.00	$7.00	
Which option would you choose? *(mark one box only)*	☐	☐	☐

Figure 1. Sample Choice Set

The first attribute in the choice experiment is Application Time. This refers to the amount of time a potential seller would have to spend to establish his eligibility to enter into a WQT contract. This time would be expended on such activities as meeting with the staff of the entity managing the market, compiling data on the field to be enrolled, and filling out paperwork. Application time would vary depending on the complexity of the program and the desires of the buyer in the contract. We set the application time to vary from 4 to 40 hours to enroll a 100-acre field, a range we assumed was large enough to capture a wide range of contract complexity.

The Monitoring Method has two categorical levels. If Monitoring Method = Annual Verification, then farmers entering into a contract would be visited at an unannounced time each year to ensure they are meeting the terms of the contract. The field where the contracted BMP is to be installed would be inspected to verify that the practice is being implemented and maintained as agreed. If Monitoring Method = Spot Check, then the farmer would be visited with a 10% probability each year, implying that one visit would occur during an average 10-year contract period. If visited, the type of inspection would be the same as with Annual Verification. These two possibilities reflect varying levels of "intrusiveness" the seller must be willing to accept.

The Penalty is a one-time fine to be paid if the seller is found in violation of the contract. Levels of this attribute range from \$50/acre to \$500/acre, a sufficiently wide variation to ensure that farmers would not find it rational to "plan on cheating" and paying the fine when caught. For example, under the Spot Check system of monitoring, the upper end of this range produces an expected penalty from cheating of \$50/acre/year. This exceeds the maximum revenue that could be earned from entering into a contract (\$25/acre/year - see below), which is also the maximum possible gain from cheating on a contract.

The BMP is the fourth attribute, which takes on four categorical levels indicating four distinct BMPs. The four BMPs vary along two dimensions. The first dimension is the type of practice – the farmer must either install a filter strip or implement no-till. The second dimension is the level of flexibility the farmer would have in meeting his contract obligations. In the case of filter strips the more flexible option would allow farmers to hay and or graze the filter-designated area. For no-till, flexibility comes in the form frequency of use – "rotational no-till" allows for some other tillage practice in 5 out of the 10 years under contract. We designed our scenarios so that Option A was always of the filter strip variety and Option B was always of the No-till variety. This reduces the number of degrees of freedom in our experimental design, by effectively reducing this four-level attribute to a two-level attribute.

The BMPs will be a significant determinant of farmers' choice if they value flexibility, or if they perceive differences in implementation costs. One complication in comparing the BMPs is that filter strips involve up-front installation costs: the land for the filter strip must be tilled, leveled, and seeded to grass in the first year. On the other hand, KSU Extension crop budgets indicate an expected cost of zero for a typical Kansas farmer to implement no-till. To make this comparison more straightforward for respondents, they were told that the installation costs of filter strips would be covered from "an outside source." This is not unrealistic, as cost share funds from both state and federal programs are available to pay for installing buffer strips statewide.

Another reason we removed the installation costs was to focus the respondent's attention on comparing the ongoing managerial costs of the practices. To clarify the managerial costs of each of these practices, farmers were given specific definitions of the practices along with a list of maintenance responsibilities. "100% No-till," for example, was defined as the tillage practice where the only equipment that breaks the soil surface is a planter, and this occurs at most once annually. For filter strips, the maintenance requirements were to regularly check for and repair any gullies that develop, to avoid using the filter strip as a roadway, and to avoid broadcast application of chemicals or manure in the filter strip area.

The final attribute is trading revenue, or the price per credit multiplied by the number of credits generated from the BMP. We varied trading revenue from $3/acre/year to $25/acre/year, following the range used by Cooper and Keim (1996) and Cooper (1997). Each BMP was assumed to generate a fixed number of credits (Table 2), and the price per credit was calculated in each scenario so that price times credits equaled the specified revenue level. For example, in Option A of the scenario shown in figure 1, our experimental design called for a revenue of $15/acre/year and a BMP of Filter Strip (with haying/grazing), a practice which would generate 6 credits/acre (Table 2).The price per credit was then calculated as $15/6 = $2.50. As described below, we generated 32 different choice sets encompassing 64 distinct trading choices. Across all 64 choices, the variation in credits (see table 2) combined with the variation in revenue ($3-$25) produced a variation in the price per credit of $0.25 to $5.00.

Table 2. Credits Generated by Best Management Practices

Best Management Practice	**Credits Generated**
	credits/acre/year
Filter strip (no haying/grazing)	12
Filter strip (with haying/grazing)	6
100% No-till	9
Rotational No-till	5

Design Procedures

As noted above, our experimental subjects were to respond to choice sets, each of which contains two trading opportunities with five attributes. Thus there are a total of ten attributes to be varied across choice sets. Our experimental design problem is to construct a collection of choice sets by systematically varying these 10 factors. 6 of these factors have 4 levels and the remaining 4 have 2 levels, implying that a complete factorial spanning all possible combinations these factors would require 65,536 distinct choice sets – obviously a prohibitive number of scenarios to present to respondents.

We used the SAS %MktRuns macro (Kuhfeld, 2005) to identify the minimum number of choice sets in an orthogonal main effects design. An orthogonal main effects design is a small sample of all combinations in the full factorial, where the chosen combinations exhibit a zero correlation among the attributes. The smallest orthogonal main effects design contains 32 choice sets, and such a design was constructed using the SAS %MktEx macro (Kuhfeld, 2005). The choice sets were then blocked into two sets of 16, so that our choice experiment came in two versions. The choice sets in our design are shown in table 3.

Table 3. Designed Choice Sets

		Option A Attributes					Option B Attributes				
Set	**Ver.**[a]	***Time***	***Monitoring*** [b]	***Penalty***	***Revenue***	***BMP*** [c]	***Time***	***Monitoring*** [b]	***Penalty***	***Revenue***	***BMP*** [d]
1	1	24	SC	50	7	FSH	4	AV	500	25	NT
2	1	4	SC	500	15	FSH	16	AV	100	15	NT
3	1	24	SC	250	15	FSNH	24	SC	500	15	RNT
4	1	40	AV	50	25	FSNH	24	AV	100	3	RNT
5	1	4	AV	500	25	FSH	4	SC	250	15	RNT
6	1	4	AV	100	3	FSNH	24	AV	250	25	NT
7	1	4	SC	250	3	FSH	40	SC	50	3	NT
8	1	24	AV	100	15	FSH	40	AV	100	7	RNT
9	1	40	SC	250	7	FSH	16	AV	250	7	RNT
10	1	40	AV	100	7	FSNH	4	SC	50	15	RNT
11	1	40	SC	50	15	FSNH	40	SC	250	3	NT
12	1	16	AV	500	3	FSNH	40	AV	500	7	RNT
13	1	24	AV	50	3	FSH	16	SC	50	25	RNT
14	1	16	AV	100	25	FSH	16	SC	500	3	NT
15	1	16	SC	250	25	FSNH	4	AV	100	25	NT
16	1	16	SC	500	7	FSNH	24	SC	50	7	NT
17	2	40	AV	250	3	FSH	4	SC	100	7	NT
18	2	4	AV	250	7	FSH	24	AV	500	3	RNT
19	2	16	AV	250	15	FSNH	16	SC	250	25	RNT
20	2	16	SC	50	3	FSH	24	SC	100	15	RNT
21	2	24	AV	250	25	FSNH	40	AV	50	15	NT
22	2	16	AV	50	7	FSH	40	AV	250	15	NT
23	2	4	AV	50	15	FSNH	4	SC	500	7	NT
24	2	24	SC	500	3	FSNH	4	AV	250	3	RNT
25	2	4	SC	100	7	FSNH	40	SC	100	25	RNT
26	2	24	AV	500	7	FSNH	16	SC	100	3	NT
27	2	24	SC	100	25	FSH	24	SC	250	7	NT
28	2	40	AV	500	15	FSH	24	AV	50	25	NT
29	2	16	SC	100	15	FSH	4	AV	50	3	RNT
30	2	40	SC	100	3	FSNH	16	AV	500	15	NT
31	2	4	SC	50	25	FSNH	16	AV	50	7	RNT
32	2	40	SC	500	25	FSH	40	SC	500	25	RNT

[a] Survey version. Sets 1-16 were in version 1; 17-32 in version 2.
[b] SC = Spot check; AV = Annual verification
[c] FSH = Filter strip (with haying/grazing); FSNH = Filter strip (no haying/grazing)
[d] NT = 100% No-till; RNT = Rotational No-till

Data Collection

Our choice experiments are being conducted in person with farmers at different producer-oriented conferences in Kansas. The first such event was already completed in conjunction with the 2006 Risk and Profit Conference, an annual event hosted by the Agricultural Economics Department at KSU. This conference was held on August 17-18, 2006 and our experiments were conducted with 39 producers in attendance. Our initial description and analysis below is based on this dataset. The second conference is a statewide Farm Bureau conference, to be held in January 2007 in Wichita. The third and fourth events are 1-day Agricultural Profitability Workshops run by KSU Extension economists, scheduled for December 2006 in northwest Kansas and for January 2007 in north-central Kansas. We plan to elicit an average of about 40 producers at each remaining conference, for a total of approximately 160 subjects.

Our data collection procedures at all these conferences are as follows. First, experimental subjects are recruited via a pre-registration mailing and an announcement at the opening conference session. The choice experiment itself is conducted during a 1-hour session, typically scheduled as a parallel session in the conference program. During this session, subjects are first shown a brief presentation on the concept of Water Quality Trading, followed by instructions to complete the choice experiments.

The instructions include much the same information as in the Design Attributes section above. A hypothetical situation was first described, in which subjects are asked to imagine that a WQT program had been developed in their region with different buyers giving them different types of opportunities to sell credits. The opportunities vary along five dimensions (the attributes in table 1). These attributes and their various levels are then explained. BMPs are explained in more detail than the other attributes, to ensure that the producers understood what their contract responsibilities would be under each. Finally, the respondents are shown an example choice set to give them practice in completing the experiment.

After allowing for clarification questions, the subjects then fill out a booklet with 16 choice sets. A printed copy of the background and instruction slides are also provided to subjects for their reference, and the instructions are also summarized at the beginning of the booklet. Each choice set in this booklet is followed by an open-ended question asking, “Why did you make this choice?” As explained in more detail below, these qualitative responses are among the first data items we are analyzing and are proving to be helpful in choosing our econometric specification. After completing the booklet each subject completes a questionnaire eliciting information on his/her farm operation, his/her attitudes toward water quality issues and policies, and demographic data. Copies of all materials used in these sessions are available from the authors.

After the instruments have been completed, each subject is paid an honorarium of $50 in cash. This is announced in the pre-registration mailing and at the opening conference session to encourage participation. Our data collection procedures and instruments were pre-tested with a small group (12) of producers from the Great Plains about one month prior to the Risk and Profit Conference.

Preliminary Data Analysis and Econometric Specification

Only preliminary analyses of the data have been performed to date. Our work so far has been descriptive and exploratory, with the intent of validating our data collection procedures and identifying the appropriate econometric specification.

Questionnaire Data: Summary Statistics

Summary statistics from our first 39 responses to the questionnaire are in Table 4. The average farmer in this sample owns 939 acres of cropland and rents 811 acres, for an average farm size of 1,750 acres. However, the distribution of size is skewed, with a few very large operations; the maximum owned acres is 6,000 and the maximum rented acres is 5,000. These statistics are reflective of the overall distribution of farm sizes in Kansas, which has a few large farms at the upper tail of the distribution. Based on the 2002 Census of Agriculture, about 10% of all farm operations in Kansas exceed 2,000 acres (NASS).

Many of the producers in the sample currently use one or more BMPs. The most popular BMP is minimum tillage, used by 53% of respondents, while the least popular on the list was subsurface application of fertilizer, with only 21% of respondents using this practice. Notwithstanding farmers' willingness to adopt BMPs, there is a persistent gap between their awareness of conservation programs and their participation in them. For example, 100% respondents are aware of the Conservation Reserve Program, but only 53% have participated in it. The gap is particularly stark for the Environmental Quality Incentives Program (EQIP), which has an awareness rate of about 90% but a participation rate of 30%. Similarly large gaps are present for the Conservation Security Program and the Kansas Buffer Initiative. Because these programs offer incentives that match and in some cases outweigh the monetary expenses of installing BMPs, the observed participation gap is consistent with the presence of intangible costs as reviewed above.

In terms of perceptions, farmers agree with the sentiment that water quality needs to be protected and that BMPs help reduce nutrient and sediment runoff. However, the average respondent was neutral on whether Kansas water supplies are polluted. The average response was also neutral on the statements that "*Mandating BMP installation and management is unfair to producers*," and that "*Environmental legislation is often unfair to producers*." Finally, the experiment itself appeared to increase subjects' knowledge of WQT, with the self-assessed level of knowledge increasing, on average, about 1.5 points on a 5-point scale.

The demographic data from our sample suggest it is fairly representative of the larger farm population, considering our relatively small sample size to date. The average age of producers in our sample is 46, compared to a population average of 56 based on the 2002 Census of Agriculture (NASS). Similarly, about 17% of our respondents were female, compared to 9% of primary farm operators in Kansas. Our sample is somewhat younger with a higher proportion of female respondents, although these may be small sample properties. If the final sample is skewed toward certain demographic cohorts, this can be corrected by assigning appropriate weights in our regression analysis.

Table 4. Summary Statistics of Initial Questionnaire Data

Item	Mean	Standard Deviation	Minimum	Maximum
Farm Characteristics				
Owned cropland (acres)	939	1602	0	6000
Rented cropland (acres)	811	1308	0	5000
Cropland bordering waterbodies (proportion)[a]	0.676	0.475	0	1
Best Management practices in use (proportion)[a]				
Filter strip	0.289	0.460	0	1
Minimum tillage	0.526	0.506	0	1
Rotational no-till	0.395	0.547	0	1
Exclusive (100%) No-till	0.289	0.460	0	1
Terraces	0.553	0.504	0	1
Sub-surface application of fertilizer	0.211	0.413	0	1
Contour farming	0.316	0.471	0	1
Familiarity/participation with conservation programs (proportion)[a]				
Conservation Reserve Program: Familiar With?	1.000	0.000	1	1
Conservation Reserve Program: Participated In?	0.526	0.506	0	1
Environmental Quality Incentives Program: Familiar With?	0.895	0.311	0	1
Environmental Quality Incentives Program: Participated In?	0.289	0.460	0	1
Conservation Security Program: Familiar With?	0.658	0.481	0	1
Conservation Security Program: Participated In?	0.081	0.277	0	1
Kansas Buffer Initiative: Familiar With?	0.421	0.500	0	1
Kansas Buffer Initiative: Participated In?	0.079	0.273	0	1
Perceptions				
Level of agreement with the following statements:[b]				
"Best management practices (BMPs) reduce nutrient and sediment runoff."	1.21	0.66	-1	2
"Kansas surface water quality needs to be protected."	1.37	0.49	1	2
"Kansas groundwater quality needs to be protected."	1.37	0.54	0	2
"Mandating BMP installation and management is unfair to producers."	0.16	1.01	-2	2
"Environmental legislation is often unfair to producers."	0.47	0.89	-1	2
"Kansas surface waters are polluted."	0.29	0.87	-2	2
"Kansas groundwater supplies are polluted."	-0.05	0.78	-2	1
Self-assessment of knowledge of Water Quality Trading:[c]				
Before participating in experiment	-1.03	1.10	-2	2
After participating in experiment	0.47	0.80	-1	2
Demographics				
Gender (1=male, 0=female)	0.834	0.374	0	1
Age (years)	45.8	12.4	23	69
Occupation				
Farmer/rancher	0.579	0.500	0	1
Landowner not actively farming	0.053	0.226	0	1
Land manager	0.053	0.226	0	1
Lender/farm advisor/educator	0.474	0.506	0	1
Farming primary occupation	0.444	0.504	0	1

[a] Responses in proportions indicate the share of subjects choosing a particular response, not a share of acreage.

[b] Responses measured on a 5-point scale, where -2=strongly disagree, -1=disagree, 0=neutral, 1=agree, and 2=strongly agree.

[b] Responses measured on a 5-point scale, where -2=very low, -1=low, 0=moderate, 1=high, and 2=very high.

Choice Data

Turning now to the choice experiments, we recorded the choice made in 16 distinct scenarios by 39 subjects, producing a dataset with 620 usable observations. Figure 2 shows the composition of these data across the 3 choices (options A, B, C) for all 39 subjects. Subjects in the figure are sorted by their frequency of choosing option C, the “do not participate” alternative. All 39 subjects chose to participate in the program (i.e., selecting either option A or B) in at least one scenario, and four subjects chose to participate in all 16 scenarios.

Participation was not dominated by either filter strip (option A) or no-till (option B) contracts. In scenarios where they participated, all but six subjects stated a willingness to choose either option, switching between the two as the non-BMP attributes (application time, monitoring, etc.) varied. In particular, only three subjects (#9, #25, #37) never chose option A and three additional subjects (#22, #26, #39) never chose B. In our entire 620-observation dataset, the distribution across the three choices are: A – 235 (38%), B – 205 (33%), and C – 180 (29%).

On the whole, these preliminary analyses indicate a quite balanced dataset across the three alternatives. This property is one way of validating the ranges of the non-BMP attributes: these attributes were varied widely enough to entice participation in both types of BMP contracts, but also led to nonparticipation in some cases. Balance is also important because we will employ a discrete choice econometric model for analysis – a model family known to be unstable and to predict poorly if the dataset is unbalanced across choices.

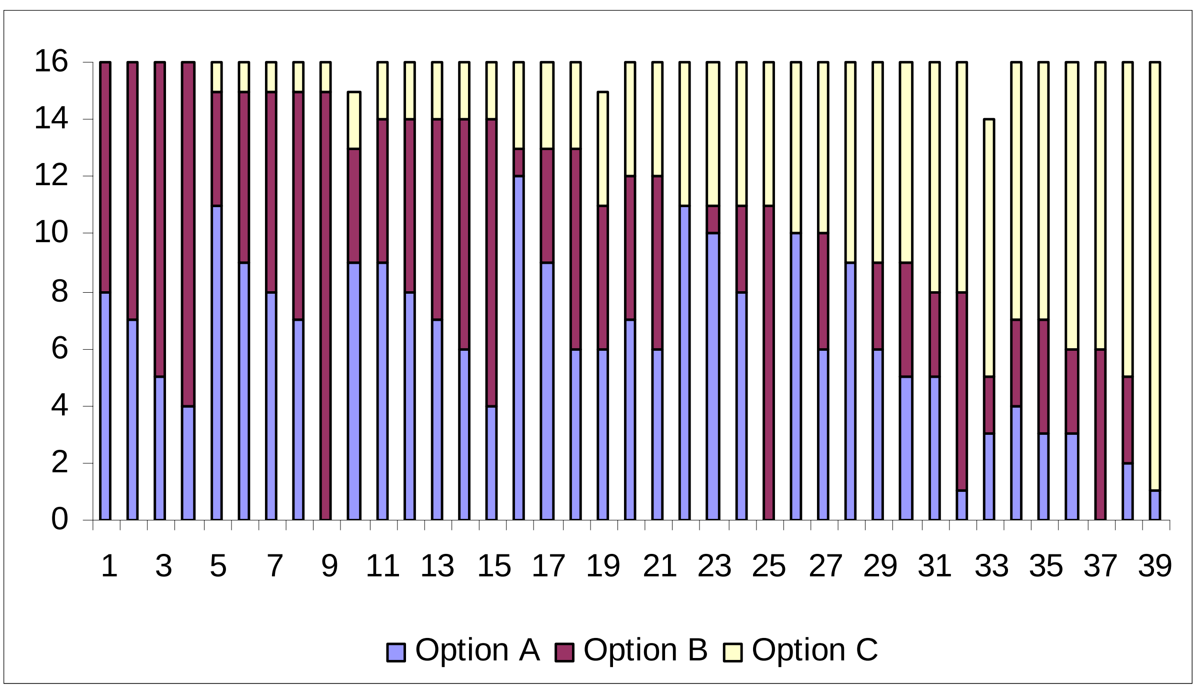

Figure 2. Distribution of Responses from Choice Experiments, First 39 Subjects

Qualitative Data: Insights for Econometric Specification

Various discrete choice econometric methods have been used to analyze choice experiment data, but all these methods are motivated by the random utility model. Suppose that on occasion t, individual i must chose one of several alternatives indexed by j. Let U_{ijt} denote the utility enjoyed by individual i if he chooses alternative j on occasion t. The random utility model posits that U_{ijt} can be partitioned into two additive components:

$$U_{ijt} = V_{ijt} + \varepsilon_{ijt},$$

where (dropping subscripts for simplicity), V is a function of observable variables and ε is a function of unobservable variables. Although individual i knows the values of both V and ε, the researcher lacks data on ε. This introduces a random element in utility across individuals from the researcher's point of view.

An estimable econometric model is developed from the random utility model by (*a*) assuming that individuals make choices to maximize utility, U, (*b*) specifying V as a function of a vector of observable variables, $\mathbf{x}$, and (*c*) making a specific distributional assumption about ε. For example, if V is specified as the linear function $V = \boldsymbol{\beta}'\mathbf{x}$ and ε follows an extreme value type II distribution then the probability that i chooses j at time t is

$$P_{ijt} = \Pr\{U_{ijt} > U_{ikt} \text{ all } k \neq j\} = \frac{\exp(\boldsymbol{\beta}'\mathbf{x}_{ijt})}{\exp(\boldsymbol{\beta}'\mathbf{x}_{ijt}) + \sum_{k \neq j} \exp(\boldsymbol{\beta}'\mathbf{x}_{ikt})}$$

This is known as the conditional logit model and is widely used in the literature. Given data on actual choices by sample of individuals, estimation of the parameters $\boldsymbol{\beta}$ can be achieved via maximum likelihood (Greene, 2003).

One assumption embedded in the conditional logit model is that the parameters, $\boldsymbol{\beta}$, are invariant across individuals. In our context, the variables in $\mathbf{x}$ would include the attributes of the various trading choices. The $\boldsymbol{\beta}$ parameters can be interpreted as the marginal utilities of these attributes, so that the conditional logit model would assume the marginal utility of each attribute is identical across subjects.

However, the qualitative data collected in our choice experiment survey directly contradict this assumption. For example, in their written follow-up responses to scenarios where one of the alternatives had a much higher Penalty than the other, different subjects provided different types of comments. One variety is well summarized by the response, "*I am assuming that I am going to comply and so I am not concerned with the penalty*." These individuals chose the option with the higher penalty, based on other attributes they found attractive such as higher revenue. Other subjects, who did not select the high penalty option, made comments similar to the following: "*Payment is great per acre … but penalty is very high and checked every year. Sure I probably would not violate but don't want to take the chance.*" Here, the concern appeared to be that the farmer would be found in violation of the contract even though he *intends* to comply.

These responses lead us to hypothesize that farmers have differing with respect to our key attributes. For the Penalty attribute, the heterogeneity in preferences would arise from differences in farmers' subjective probabilities of being found in violation when intending to comply, as well as differences in their risk preferences. In order to test this hypothesis, we must specify a model that allows the β parameters to differ across individuals. One such model is the random parameters logit model. One or more of the parameters in the β vector are assumed to have a distribution across individuals, which can be specified by the researcher (e.g., normal or log-normal distribution). Rather than estimating the values of the β's per se, the econometric problem is to estimate the underlying distributional parameters of the randomly specified β's across people (e.g., means, variances, and covariances). This model will be pursued to formally test whether the marginal utility parameters differ across farmers.

Concluding Remarks and Next Steps

The econometric model to be estimated from the choice data will be capable of predicting the trading choices of farmers in a WQT program under different trading rules. As part of our ongoing research project, our next goal is to run trading simulations under different types of rules to assess their effect on market performance. These simulations will be accomplished by inserting our estimated equations into a trading simulation model already developed by Smith (2004), which in turn is based on the sequential bilateral trading algorithm of Atkinson and Tietenberg.

Once the trading simulation model is complete, it will be linked to a biophysical watershed model being developed for the Kansas/Delaware Subbasin using SWAT (Arnold et al., 1998; Neitsch et al., 2001). The linked models will then be run in tandem to assess the joint performance of various market designs on economic measures as well as on water quality in different river segments. The objective is to identify a set of trading rules that are simple enough to attract adequate participation while being sufficiently tailored to ensure that water quality goals are indeed met.

As this project is a work in progress and data collection is still underway, only very preliminary results are available. The initial results obtained from our choice experiments suggest that the attribute levels provide a range of incentives to which subjects respond in different ways. Demographic variables in our dataset suggest our sample is so far weighted somewhat toward younger and female producers. More formal tests of demographic representativeness will be conduced as data collection progresses, and adjustments will be made as needed to change our sampling strategy or correct our regression by reweighting different demographic cohorts.

References

Adamowicz. W. et al. "Perceptions versus Objective Measures of Environmental Quality in Combined Revealed and Stated Preference Models of Environmental Valuation." *Journal of Environmental Economics and Management* 32(1997): 65-84.

Arnold, J.G., R. Srinivasan, R.S. Muttiah, J.R. Williams. "Large Area Hydrologic Modeling and Assessment, Part I: Model Development." *J. American Water Resources Association* 34(1998): 73-89.

Atkinson, S., and T. Tietenberg. "Market Failure in Incentive-Based Regulation: The Case of Emissions Trading." *Journal of Environmental Economics and Management* 21(1991): 17-31.

Breetz, H.L., et al. *Water Quality Trading and Offset Initiatives in the United States: A Comprehensive Survey.* Report for the EPA. Hanover, NH: Dartmouth College Rockefeller Center, 2004.

Cooper, J. "Combining Actual and Contingent Behavior Data to Model Farmer Adoption of Water Quality Protection Practices." *Journal of Agricultural and Resource Economics* 22(1997): 30-43

Cooper, J, and R. Keim. "Incentive Payments to Encourage Farmer Adoption of Water Quality Protection Practices." *American Journal of Agricultural Economics* 78(1996): 54-64.

Earnhart, D. "Combining Revealed and Stated Preference Methods to Value Environmental Amenities at Residential Locations." *Land Economics* 77(2001): 12-29.

Fox, J.A., D.J. Hayes, and J.F. Shogren. "Consumer Preferences for Food Irradiation: How Favorable and Unfavorable Descriptions Affect Preferences for Irradiated Pork in Experimental Auctions." *Journal of Risk and Uncertainty*, 24(2002):75-95.

Faeth, Paul. 2000. *Fertile Ground: Nutrient Trading's Potential to Cost-Effectively Improve Water Quality*. Washington, D.C.: World Resources Institute. Available at: www.wri.org/water.nutrient.html

.
Greene, W. *Econometric Analysis*, 5th Edition. New York: Prentice Hall, 2003.

Hoag, D.L., and J.S. Hughes-Popp. "Theory and Practice of Pollution Credit Trading." *Review of* Agricultural *Economics* 19(1997): 252-262.

Kuhfeld, W. *Marketing Research Methods in SAS*. TS-722. Cary, NC: SAS Institute, 2005. Available at http://support.sas.com/techsup/tnote/tnote_stat.html#market.

National Agricultural Statistics Service (NASS). "2002 Census of Agriculture." http://www.nass.usda.gov/Census_of_Agriculture/index.asp.

National Center for Environmental Economics (NCEE). “The United States Experience with Economic Incentives for Protecting the Environment.” Publication EPA-240-R-01-001. Washington, D.C.: U.S. Environmental Protection Agency, 2001

Neitsch, S.L., J.G. Arnold, J.R. Kiniry, J.R. Williams. *Soil and Water Assessment Tool* Theoretical *Documentation, Version 2000*. Texas Agricultural Experiment Station, Temple, TX, 2001.

Smith, C. “A Water Quality Trading Simulation for Northeast Kansas.” Unpublished M.S. Thesis, Department of Agricultural Economics, Kansas State University, Manhattan Kansas, 2004.

Incorporating Wetlands in Water Quality Trading Programs: Economic and Ecological Considerations

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Abstract

Water quality trading has grown in popularity and scope in recent years owing to its potential as a flexible low cost way to achieve water quality goals, especially nutrient removal goals. Wetlands provide a variety of ecosystem services. They can sequester CO_2, they can provide habitat and biodiversity and they can remove nitrogen from nonpoint sources of pollution before they enter receiving streams or rivers. Because of this last service regulators are interested in determining the best way to allow traders, primarily agricultural traders, to use restored and protected wetlands in a water quality trading policy. Key to the problem is the existence of the ancillary benefits of wetlands. In this paper we examine the options of 1) including the ancillary benefits of a properly functioning wetland in the market for nutrient removal through subsidies and unique trading ratios, or 2) allowing a producer to trade the various services offered by wetlands in various markets. We also examine which option might be preferred depending on the shape of the marginal benefits curve.

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Introduction

Though great successes are credited to the air quality trading programs, water quality trading has proved problematic. Authors like King and Kuch (2003) find that there are both supply-side and demand-side obstacles to trading. For example, water quality trading programs that control nutrients competed with "green payments" for reducing nonpoint sources of nutrient pollution. Green payments, such as the Conservation Reserve Program, Wetlands Reserve Program, etc., provide assistance to landowners to address environmental issues like soil erosion and damaged or lost wetlands and habitat. Thus, these activities reduced the potential supply of water quality trading credits. Point sources, or potential credit demanders, find the idea of trading with nonpoint sources inequitable given the existing subsidy or green payment programs. Perhaps the largest obstacle facing water quality trading is the fact that the markets are too small to take advantage of those things markets do well.

To this end it has been suggested by many that increasing the size of the market for nutrient trading through the inclusion of wetlands, which act as a nutrient reduction technology, will increase the size of the market enough to bring about a successful program (Raffini and Robertson 2005). There are other benefits from using wetlands that make them attractive. Wetlands sequester CO_2, and wetlands provide habitat. There exist markets for the other two services provided by wetlands. Should wetlands be incorporated in the water quality trading market through the use of trading ratios and subsidies? Or should those who do restore or create wetlands for the purpose of nutrient reduction be able to sell the other services on other markets? The answer depends on a variety of legal, economic, and ecological factors (Heberling et al. 2007).

Austin et al. (1997) and Feng and Kling (2005) both study the ancillary benefits in pollution trading markets. Austin et al. (1997) examine the cost-effective allocation when NOx emissions affected both air quality and water quality. The constraint was water quality standards for the Chesapeake Bay; therefore, the air benefits were ancillary. Feng and Kling (2005) consider the cobenefits of planting practices that sequester carbon. Certain sequestration activities, like planting cover crops or changing tillage practices, also reduce soil erosion and runoff or improve water quality. The reduction in erosion or improved water quality is an externality to the carbon market. Both papers focus on the ancillary benefits of reducing the particular pollutant when they model their problems, which is slightly different from the issue we address here. When using wetlands in water quality trading programs, it is not the reduction in the pollutant that “co-causes” the ancillary benefit; rather, it is the abatement activity or specific technology itself that creates the ancillary benefits.

This paper proceeds as follows, first we discuss the nutrient removal capacity of wetlands and the ancillary ecological and economic benefits they provide. Then we look at some of the economics literature on the theory underlying the choice to use one market or allow the use of multiple markets. Next, we look specifically at the ecology of the wetland system in an effort to appropriately characterize its creation of benefits. Several authors look at the costs and benefits of reducing nutrient loading to the Gulf of Mexico in “The summary evaluation of the economic costs and benefits of methods for reducing nutrient loads to the Gulf of Mexico” (Doering et al. 1999). In that paper there is a brief discussion of price vs. quantity controls, a la Weitzman (1974), focusing on the relative slopes of the marginal cost and marginal benefits curves. It has been suggested elsewhere in the literature (e.g., Woodward and Han 2004, Montero 2001) that this might be not simply be an interesting secondary focus, but rather an integral policy-guiding feature of the problem. It may dictate whether we prefer a separate market for the ancillary benefits of wetlands or use subsidies and unique trading ratios to encourage their creation.

Wetlands and Water Quality Trading Markets

The basic requirements for a well functioning transferable permit market have been outlined numerous times (see, for example, Heal 2000, Godard 2001, and Biller 2003). These requirements include such things as clear, transferable property rights, bankable permits, securitization, adequate information about damages, legal cap or limit, defensible initial allocation of permits or rights, heterogeneity in ability or cost of control and damage, and a large number of participants. It has also been outlined in many places how water quality trading schemes around the country are not living up to their hype, and there is a general feeling that limited participation or "thin markets" are the primary culprit (King and Kuch 2003). There are few opportunities for traders to realize the full potential of the market, robust and efficient trades are seldom seen. Allowing the use of wetlands in water quality trading programs serves many purposes including increasing the size of the market and increasing the acres of wetlands.

Assuming we can find watersheds where the supply-side and demand-side obstacles are minimized, why do we need to specifically discuss wetlands and trading markets? If wetlands were, in most respects, similar to other nutrient abatement technology, no further discussion would be needed. Producers would choose from a suite of available abatement technologies based on minimizing their costs and would choose wetlands if they represented the least cost method of creating nutrient credits.

Unlike some other types of abatement technology though, wetlands have other functions that benefit humans directly or indirectly. Economists refer to these functions as wetland services; wetlands may control nutrients, and they may also produce habitat for birds, control flooding, and reduce sediments. The wetlands could be restored or constructed for the purpose of nutrient abatement and could also create these additional services. Some individuals who are not involved in the water quality trading transaction could benefit from these wetland services, but they would not have to pay for them. U.S. EPA considers them ancillary benefits of water quality trading which could accrue to the general public or just to the landowner.

Byström (1998) examines the abatement costs of using wetlands to control nutrients. He suggests that the social benefits could substantially lower the abatement costs of using wetlands, but he does not explicitly estimate these costs.

Ribaudo et al.(2001) looks at reducing nitrogen in the Mississippi Basin through fertilizer reduction or wetland restoration. Not only did Ribaudo et al. include the private costs, social benefits, such as erosion benefits and wetland benefits (e.g., $550/acre), were also incorporated in their model. They find that the social marginal costs of control using wetlands become lower after about 1250 tonnes of nitrogen reduction which occurs when the marginal cost of control of fertilizer reduction catches up to the opportunity costs of land.

Regardless of the wetland functions, economic theory suggests that the producer will not consider the ancillary benefits (positive externality of producing wetlands) because the benefits do not enter the profit maximising decision. If the externality were internalized, then, and only then, would the producer face the social (net) costs. What regulators need to determine is whether the ancillary benefits actually should play a role in the decision of the credit producer (i.e., should regulators ignore the benefits of wetlands and allow producers to minimize their costs of reducing nutrients and sediments?).

Proposition 1: Command-and-Control

Regulators could require all producers of pollution credits to build wetlands to abate nutrients. This, in effect, takes the decision out of the hands of the producers. There would likely be situations where wetlands are not the least cost option and requiring the use of wetlands would not be cost-effective (e.g., limited land space and increasing opportunity costs). Forcing a particular abatement technology goes back to the problem with command-and-control policies which are rarely cost-effective. If regulators decide these benefits should be considered, and U.S. EPA's Water Quality Trading Policy suggests they should, are there other approaches that make economic sense?

Model

For this paper, we assume that the water quality trading market is the primary market and follow the model presented in Horan and Shortle (2005). They focus on a trading program based on expected loadings for the nonpoint source (rather than on inputs). The model assumes a single point source (e.g., a municipal separate storm sewer system (MS4)) and a single nonpoint source (e.g., a farm) for a watershed.

Emissions for the point source, e, are controlled with certainty and known costs c(e). The nonpoint source emissions are considered random, $r(\mathbf{x}, \theta)$, with jth element of $\mathbf{x}$ (a mx1 vector) representing the set of production decisions related to the technology for production and pollution control. The random variable, θ, represents stochastic events that affect runoff, like weather. We assume the nonpoint source profit depends on the choice of $\mathbf{x}$ and the difference between the profits with no regulations ($\mathbf{x_0}$) and profits under regulations ($\mathbf{x}$) is the nonpoint source pollution control costs, $c_r(x) = \pi(\mathbf{x_0})-\pi(\mathbf{x})$. Pollution from each source causes damage costs, D(e, r) and social costs are then $TC=c(e) + c_r(\mathbf{x}) + E\{D(e, r)\}$.

We assume that some pollution abatement technology provides benefits to third parties outside of the market; an additional component representing ancillary benefits is needed. This assumption differs from Austin et al. (1997) and Feng and Kling (2005), who model the benefits as a function of the reduction of the pollutant, not the technology. Therefore, total ancillary benefits are $B[x_j]$; however, $B[x_j]>0$ only when j=w where w represents a specific pollution control technology that affects individuals outside of the market.[1] We assume $B[x_j]$ is known with certainty for this paper, but we understand that this is an oversimplification. The benefit function is twice continuously differentiable, increasing in x_j ($B'[x_j]>0$) and exhibits decreasing marginal returns ($B''[x_j]<0$).

An *ex ante* efficient allocation of pollution control minimizes the total social cost (TSC): $TSC=c_e(e) + c_r(\mathbf{x}) + E\{D(e, r)\}- B[x_j]$. The necessary conditions are:

$$(1)\ \frac{\partial TSC}{\partial e} = c^{'}(e) + E\left\{\frac{\partial D}{\partial e}\right\} = 0$$

$$(2)\ \frac{\partial TSC}{\partial x_j} = \frac{\partial c_r(\mathbf{x})}{\partial x_j} + E\left\{\frac{\partial D}{\partial r}\frac{\partial r}{\partial x_j}\right\} = 0, \forall j \neq w$$

In the case at hand however, since the abatement technology employed by the nonpoint source leads to ancillary benefits, the marginal external benefit is included in the necessary condition:

[1] For this model, we acknowledge that some abatement technology could lead to external costs. However, for this application, we assume only external benefits; we ignore the case too of non-convexities in the production of the various benefits.

$$(3)\ \frac{\partial TSC}{\partial x_j} = \frac{\partial c_r(\mathbf{x})}{\partial x_j} + E\left\{\frac{\partial D}{\partial r}\frac{\partial r}{\partial x_j}\right\} - B'(x_j) = 0, j = w$$

For all x_j, where $j \neq w$, the marginal expected damages should equal the marginal private benefit for using the input and the marginal private costs of using abatement technology should equal the expected reduction in marginal damage costs. When j=w, the marginal private costs should equal the total of the expected reduction marginal damage costs and the marginal external benefits from the technology.

Market Equilibrium

Following existing trading markets, we use two sets of permits: point source, $\hat{e}$, and nonpoint source, $\hat{r}$. The MS4 must have a mix of these permits at least equal to their emissions. A trading ratio, t, equates emissions to expected loadings: $t = \left|\frac{d\hat{r}}{d\hat{e}}\right|$.

The MS4 will choose emission levels that minimize costs, given price p for emission permits and price q for expected loadings permits to minimize costs, $C=c(e) + q[\hat{e}_{ps} - \hat{e}^0_{ps}] + p[\hat{r}_{ps} - \hat{r}^0_{ps}]$, where superscript 0 represents the initial holdings of permits. It faces the constraint that emissions cannot be greater than the permits it holds, $e \leq \hat{e}_{ps} + (1/t)\hat{r}_{ps}$, where (1/t) is the trading ratio to convert nonpoint source permits to emissions. Assuming that the constraint is satisfied as an equality and assuming the initial allocation of nonpoint source permits for the MS4 is zero, we can substitute the constraint into the cost function. First order conditions remain unchanged from Horan and Shortle (2005). We learn that the trading ratio at the margin is t=q/p and the MS4's costs can be simplified to $C=c(e) + q[e - \hat{e}^0_{ps}]$.

The nonpoint source also has to meet conditions for a market solution. The social cost function for the nonpoint source is $P=c_r(\mathbf{x}) + q[\hat{e}_{nps} - \hat{e}^0_{nps}] + p[\hat{r}_{nps} - \hat{r}^0_{nps}] - B[x_j]$ which is defined similarly as above. We assume that the nonpoint source does not hold any point source permits initially and it faces a loadings constraint, $r \leq \hat{r}_{nps} + t\hat{e}_{nps}$, where t is the trading ratio. Assuming the constraint is met as an equality, we can rewrite social costs as $P=c_r(\mathbf{x}) + p[E[r(\mathbf{x}, \theta)] - \hat{r}^0_{nps}] - B[x_j]$. The first order condition for optimal input use is

$$\text{(4)} \quad \frac{\partial P}{\partial x_j} = \frac{\partial c_r(\mathbf{x})}{\partial x_j} + pE\left[\frac{\partial r}{\partial x_j}\right] - B'[x_j] = 0, j = w$$

Finally, we know that for the market to clear, we need to have more permits than emissions and expected loadings:

$$\text{(5)} \quad \hat{e}^0 + \left(\frac{1}{t}\right)\hat{r}^0 \geq e + \left(\frac{1}{t}\right)E[r(x_j, \theta)].$$

By basing the number of permits allocated and trading ratios on the results above, we can create the optimal water quality trading program. However, an optimal trading program is not realistic, and Horan and Shortle (2005) present a "conditionally optimal" trading program, based on an environmental authority choosing the number of emission permits available for the market.

We follow their approach for determining the prices and conditionally optimal trading ratio, but allow for the inclusion of at least one of the recognized ancillary benefits of wetlands as the abatement technology. We substitute the derived demands x(p) and e(q) into the total social cost function subject to the market clearing constraint . The Lagrangian is

$$\text{(6)} \quad L = c(e(q) + c_r(x(p)) + E[D(e(q), r(x(p), \theta))] - B[x(p)] + \lambda[(\hat{e}^0 - e(q)) + \left(\frac{p}{q}\right)[\hat{r}^0 - E(r(x(p), \theta))]]$$

where lambda equals the shadow value of increased permit numbers.

The important necessary conditions when ancillary benefits are produced are the market clearing constraint and

$$(7)\quad \frac{\partial L}{\partial p}=\frac{\partial c_r(\mathbf{x})}{\partial x_j}\frac{dx_j}{dp}+E\left[\frac{\partial D}{\partial r}\frac{dr}{dx_j}\right]\frac{dx_j}{dp}+\lambda\frac{1}{q}(\hat{r}^0-E[r])-\lambda\frac{p}{q}E\left[\frac{dr}{dx_j}\right]\frac{dx_j}{dp}-B'[x_j]=0,\forall j=w$$

Based on equation (7), an increase in p has two effects: it leads to a decrease in input use and expected loadings and it decreases the trading ratio, t. The first two right-hand terms show that a decrease in emissions increases abatement costs and decreases expected damages. The fourth term shows that, at the margin, decreasing expected loadings will have a social cost given the constraint.

From equation (7), we can estimate the conditionally optimal price for the expected loadings permit and the conditionally optimal trading ratio. The basic results for the conditionally optimal emission permit price are taken from Horan and Shortle (2005), but with the inclusion of a term that captures ancillary benefits which has interesting implications for the nonpoint source permits and trading ratio. Suppose the only change a nonpoint source makes on the land is adding wetlands for controlling nutrients and it creates additional habitat for wildlife. Substituting the necessary condition for the nonpoint source from the market equilibrium and the estimate of the trading ratio into equation (7), we can solve for p:

$$(8)\quad p = \frac{E\left[\frac{\partial D}{\partial r}\frac{dr}{dx_j}\right]}{E\left[\frac{\partial r}{\partial x_j}\right]} + \frac{\lambda \frac{1}{q}(\hat{r}^0 - E[r])}{E\left[\frac{\partial r}{\partial x_j}\right]}\frac{dp}{dx_j} - \lambda\frac{p}{q} - \frac{B'[x_j]}{E\left[\frac{\partial r}{\partial x_j}\right]}$$

$$= E\left[\frac{\partial D}{\partial r}\right] + \frac{\operatorname{cov}\left(\frac{\partial D}{\partial r}, \frac{\partial r}{\partial x_j}\right) - B'[x_j]}{E\left[\frac{\partial r}{\partial x_j}\right]} - \lambda\frac{p}{q} + \lambda\frac{p}{q}\left(\frac{(\hat{r}^0 - E[r])}{E[r]}\right)\varepsilon_{pr}, \forall j = w$$

where $\varepsilon_{pr}<0$ is the nonpoint source's inverse elasticity of demand for expected pollution loads. With no ancillary benefits, the marginal external benefits drops out and the price of the expected loadings permit is the same as Horan and Shortle (2005). With ancillary benefits, the change in price depends on the sign of the covariance. A negative sign suggests that expected loadings permit price should be higher when ancillary benefits are created. With a positive sign, the change in price depends on whether $\operatorname{cov}(\partial D/\partial r, \partial r/\partial x_j)$ is greater, less than, or equal to $B'[x_j]$.

Malik et al. (1993), Shortle (1987), and Horan and Shortle (2005) discuss the sign of the covariance term. If the damage function is convex in r, then the covariance term has the same sign as $\operatorname{cov}(r, E[\partial r/\partial x_j])$. The sign of this equals the change in the variance of nonpoint source pollution given a change in the level of abatement. If the level of abatement decreases the variance of nonpoint source pollution, then the covariance is negative. While one would think that increasing the level of a specific abatement technology would always reduce the variance of the targeted pollution this is not necessarily the case in such complex systems as wetlands. Bÿstrom et al. (2000) and Mitsch and Gosselink (2000) indicate that wetlands are able to reduce the variance of the nonpoint source pollution. If true, the covariance term is negative and price should be higher when ancillary benefits are generated. But evidence from constructed wetlands in Ohio gathered by Spieles and Mitsch (2000) points to a possible increase in variance in a high nutrient riverine system, which means the covariance term is positive. And Moustafa et al. (1996) find in a wetland in south Florida covariance for abatement of Phosphorous decreased but that for Nitrogen did not, further highlighting the complexity of these systems.

Continuing our assumption that wetlands are created on the nonpoint source land, we can try to understand how the additional benefits affect the trading ratio. Knowing that the trading ratio is the ratio of permit prices, we can use the results above for p and q and develop a trading ratio when ancillary benefits are generated:

$$(9) \quad t = \frac{E\left[\frac{\partial D}{\partial e}\right] - \lambda\left(\frac{(e-\hat{e}^0)}{e}\right)\varepsilon_{qe} - \lambda\left(\frac{(\hat{r}^0 - E[r])}{E[r]}\right)\varepsilon_{pr}}{E\left[\frac{\partial D}{\partial r}\right] + \left[\frac{\operatorname{cov}\left(\frac{\partial D}{\partial r}, \frac{\partial r}{\partial x_j}\right) - B'[x_j]}{E\left[\frac{\partial r}{\partial x_j}\right]}\right]}$$

If nonpoint loadings are known, no ancillary benefits are produced, and the number of permits is set optimally, the trading ratio reduces to the ratio of damage impacts from emissions and loadings. Incorporating stochastic nonpoint loadings adds the second term in the denominator, what Malik et al. call the "marginal damage premium." It becomes apparent for this trading ratio that the sign of the marginal damage premium depends on the covariance term. When ancillary benefits occur, the marginal damage premium includes the marginal external benefit. The sign for the marginal benefit is assumed positive. We assume that the loading function is decreasing in x_j, meaning the denominator is negative. If the covariance is negative, the term in the large bracket is positive and the trading ratio should be smaller. With a positive sign, the trading ratio depends on whether $\operatorname{cov}(\partial D/\partial r, \partial r/\partial x_j)$ is greater than, less than, or equal to $B'[x_j]$.

When we assume multiple changes by the nonpoint source, the trading ratio becomes

$$(10) \quad t = \frac{E\left[\frac{\partial D}{\partial e}\right] - \lambda\left(\frac{(e-\hat{e}^0)}{e}\right)\varepsilon_{qe} - \lambda\left(\frac{(\hat{r}^0 - E[r])}{E[r]}\right)\varepsilon_{pr}}{E\left[\frac{\partial D}{\partial r}\right] + \left[\sum_{j=1}^{m} \frac{\operatorname{cov}\left(\frac{\partial D}{\partial r}, \frac{\partial r}{\partial x_j}\right) - B'[x_j]}{E\left[\frac{\partial r}{\partial x_j}\right]}\right]}$$

We propose two ways to internalize this positive externality: one is to provide some kind of subsidy and unique trading ratio within the program that specifically rewards the use of wetlands over other technologies and further that rewards "better" wetlands incrementally. The other way is to allow wetlands to be traded in multiple markets. That is, the nonpoint source would get credit for the creation of a wetland in the water quality trading program, and could solicit credit for the same wetland under a carbon sequestration market and if applicable a biodiversity market.

Proposition 2: Subsidy and Unique Trading Ratio

There are two principle reasons to capture the ancillary benefits of wetlands in the nutrient trading market, one is the reduction of transactions costs and the other is the increase of market size. It seems that nutrient reduction and habitat preservation occur in the same places, combining a market will increase transfer opportunities and extend the range of solutions open to agents, while reducing transaction and organization costs (Godard 2001).

Given that the expected marginal external benefits from abatement technology only enters the price of the expected loadings permits and the trading ratio, there are likely incentives within those components that might encourage the use of wetlands. If the external benefits were internalized, then the nonpoint source and point source would have additional incentives for using wetlands in a water quality trading program.

When no external benefits exist with the abatement technology, the trading ratio is that calculated by Horan and Shortle (2005). Ancillary benefits lead to either a higher or lower trading ratio depending on how wetlands affect the variance of the loadings. According to Malik et al. (1993), sources will take into account the abatement costs when conducting trades, but not costs from the variability of nonpoint source pollution. This would be similar for the ancillary benefits. Malik et al. (1993) propose that adjusting the trading ratio will help to internalize the costs.

A subsidy provided to the producer of the credit of the size $\frac{-B'(x_j)}{E(\partial r / \partial x_j)}$ is needed to correct the price of the expected loadings permit.[2] It is the appropriate subsidy to encourage the farmer to construct or restore a wetland that creates the largest ancillary benefits possible (given land and cost constraints). The subsidy does not equal the marginal benefits; it differs because we cannot measure loadings with certainty. Because loadings are estimated, the marginal benefits are adjusted depending on how the runoff function is affected by the abatement technology. In addition, a unique trading ratio would be used for demanders of credits created by wetlands. If the covariance is negative or if the covariance is positive, but smaller than the marginal benefits, then the trading ratio should be smaller. A smaller trading ratio means fewer nonpoint source permits trade for one unit of emissions, making the nonpoint source permits more attractive for the MS4. If the covariance is positive and is larger than the marginal benefits, the nonpoint source permits are not as attractive for the MS4. This means that to encourage the use of wetlands, wetlands must reduce the variance of the loadings, have a relatively large marginal benefit, or both.

[2] This still might not be enough to encourage the use of wetlands if the other abatement technology is less expensive.

Proposition 3: Multiple Markets

By allowing producers to sell different types of credits in different markets, we allow them to make decisions about their own land that will maximize their utility (or profit). Rather than having a regulator pay for the proper wetlands, we allow the markets to create the incentives of what should be bought and sold. Therefore, this does not necessarily encourage the use of wetlands if the markets do not support the production decisions. Here the point source purchases the credit of nutrient abatement and the other markets provide incentives to the nonpoint source as to how the credit should be produced. If properly designed, the externality is internalized as all additional services could be bought and sold in a market. In addition, there are probably different versions of this proposition related to how the wetlands are restored or constructed and how all the wetland services interact (e.g., substitutes or complements).

Unlike Proposition 2 where economists must estimate the value of the additional wetland services, the second proposal allows the market to value them. 'Multiple markets' refers to the producer's ability to sell different types of credits in different markets (Woodward and Han 2004, Kieser and Associates 2004, and ELI 2005). If well-functioning markets (as described above) were to exist for the different services provided by wetlands, the ancillary benefits would be accounted and the externalities would be internalized. Building wetlands might create credits for nutrient abatement, endangered species habitat, greenhouse gases, and possibly wetland mitigation banks. The services are no longer externalities of the water quality trading market as they are sold as credits in other markets. The incentive for creating wetlands, then, becomes the additional income from trading in other markets. The socially optimal level of wetlands would occur once markets exist for all relevant wetland services. If this were the case, the prices for the permits and the trading ratio would be the same as Horan and Shortle (2005). The marginal benefit term would drop out of equations (8) and (9), leading to the conditional optimum.

If the producer can sell different credits in different markets, then they may have incentive to build wetlands. Producers will react to the multiple markets and make their decisions based on their profits. Holding the number of acres constant, the producer, of course, would choose the mix of abatement technologies that produce the most money. Unlike trading ratios and subsidies, the incentives created by multiple markets are the prices received for the credits (not the value of the ancillary benefits) and the production and monitoring costs.

One market or multiple markets?

Determining which way of internalizing the externality is a difficult process and we appeal to the ecology of wetlands to determine which of these two propositions bring about the socially optimal amount of wetland use for mitigation of nutrient loading. It is clear that wetland services should not be ignored due to externalities and potential suppliers of credits should not be forced to use wetlands as a way to control nutrients because of the inefficiencies that could be created. To decide on what option makes the most sense, we turn to Montero (2001) and Woodward and Han (2004) who suggest that the decision to combine all services into the nutrient market using trading ratios and subsidies or to create multiple markets depends on the relative shape of the marginal benefits and marginal cost curves and the underlying ecological attributes. Based on Weitzman (1974), a flatter marginal benefit curve relative to the marginal cost curve suggests that multiple markets will cause a larger dead weight loss. The decision to combine or not to combine depends on the relative shape of the marginal benefits and marginal cost curves and the underlying ecological attributes.

Before looking at the marginal benefits curve, however we need to examine the graphical relationship between the creation of ancillary benefits and the reduction of nutrient loads by the wetland. If pollutants are co-produced then reduction of one will automatically reduce the other to some extent. In our example, when nutrients are reduced through creation of a wetland (of a certain quality) the ancillary benefits are increased. As the farmer spends more on the wetland such that it abates more nutrient runoff, it creates more ancillary benefits. In figure 1 we have adapted Helfand (1991) to show that as the farmer pays along $C_N =$ P in the market for nutrient reduction, he creates more ancillary benefits without it costing extra, as he does so along the zero isocost line C_b The amount of ancillary benefits, in this case we have chosen to call it "bio" or some measure of ancillary biological benefits, created will be A.

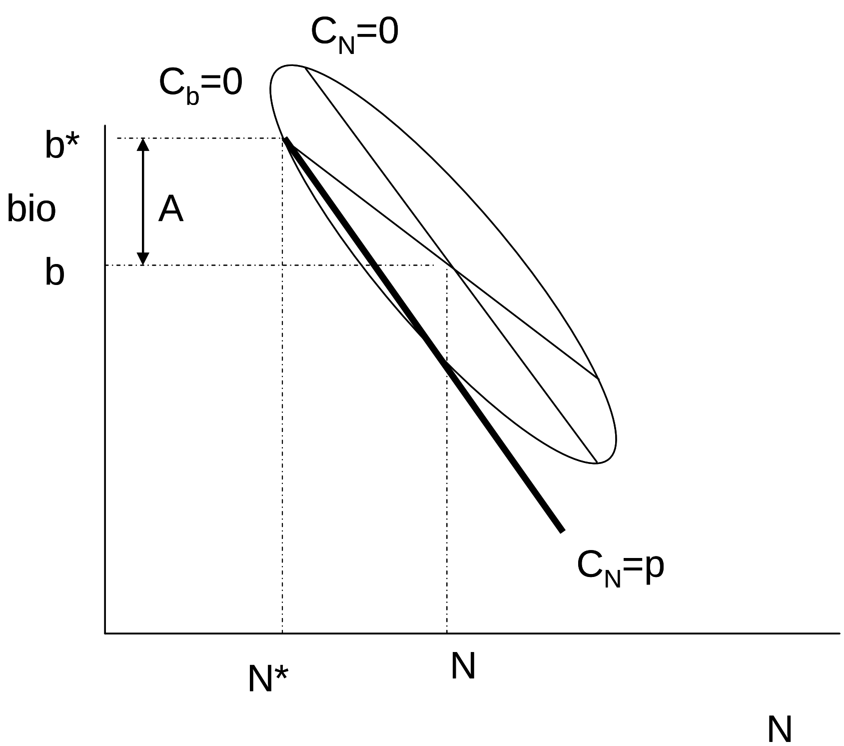

Figure 1

Woodward and Han (2004) build on that and use a figure similar to figure 1, to show that the flatter the MB curve the more deadweight loss separate trading programs will cause.

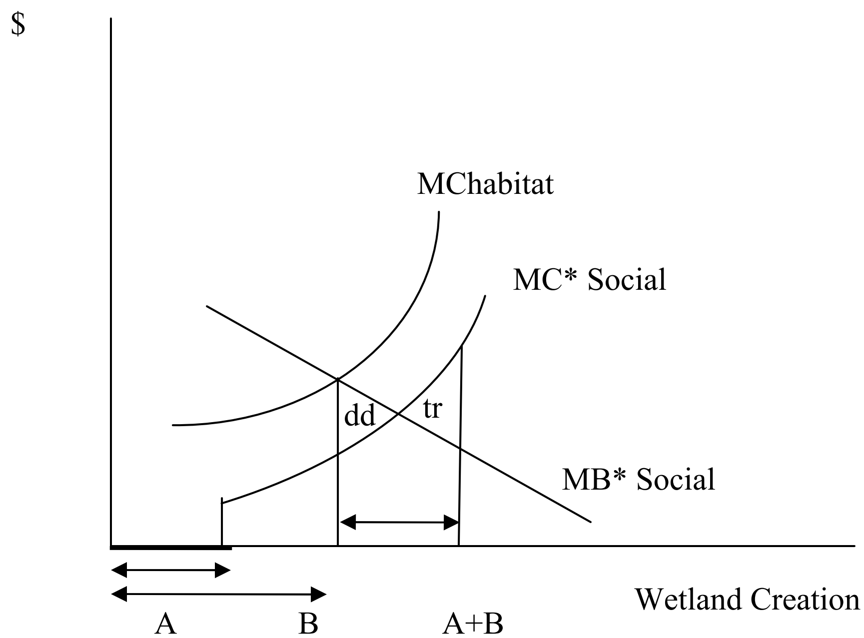

Figure 2

It becomes apparent that it is critical to have an accurate portrayal of the benefits curve. We asked three professional ecologists to quickly sketch the benefits curve for biodiversity creation from wetlands. This resulted in Figures 3, 4, and 5, which range from a simple natural log relationship to rather complex series of inflection points and threshold levels.

Our ecologists gave us total benefits curves, so note that the marginal benefits curves TB', will have similar inflections and nuances. Note also that it will be critical to accurately portray the benefits curve and to determine where on the benefits curve the proposed policy plans to operate.

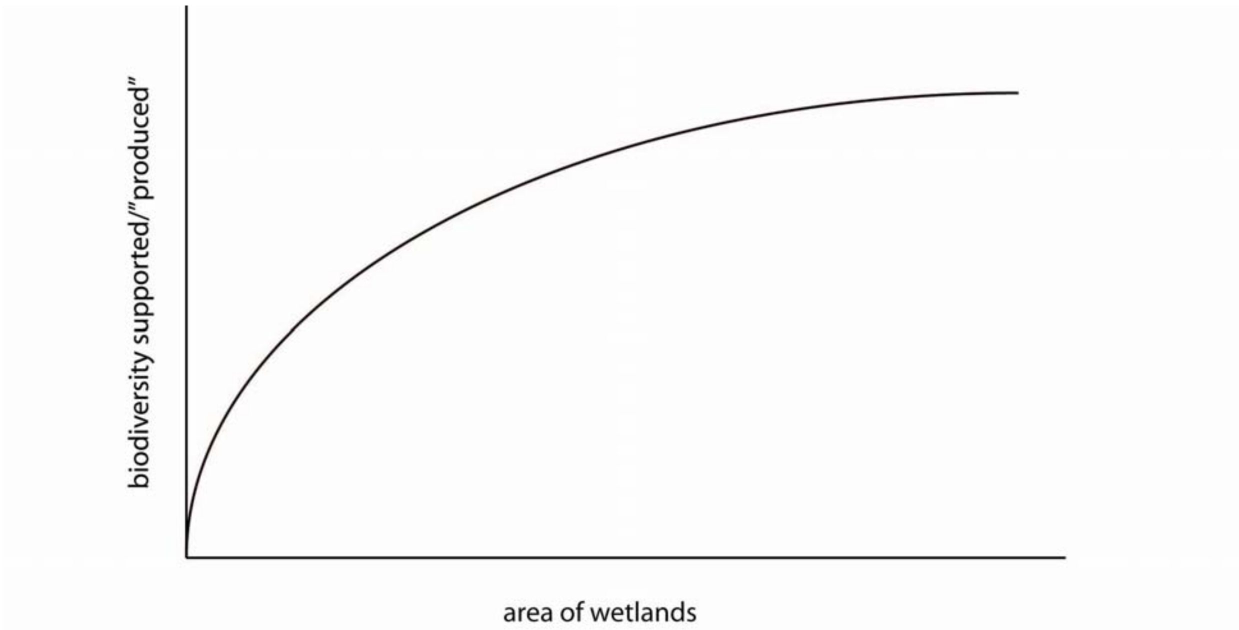

Figure 3

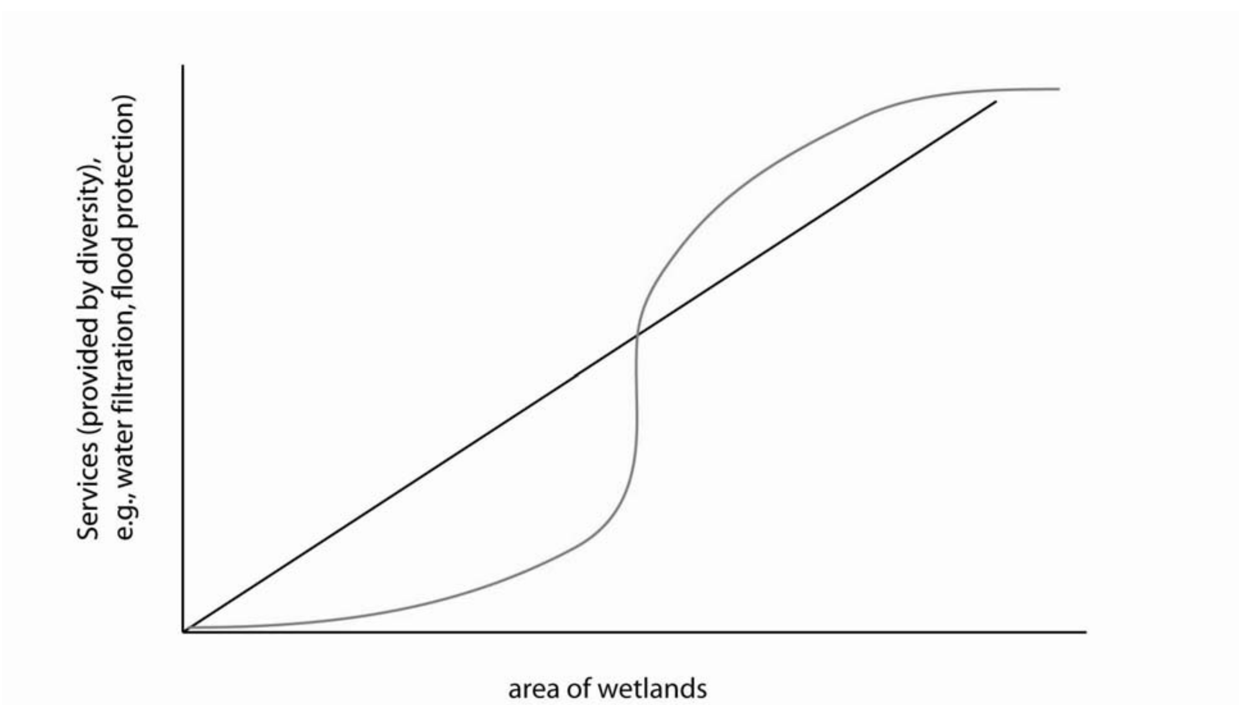

Figure 4

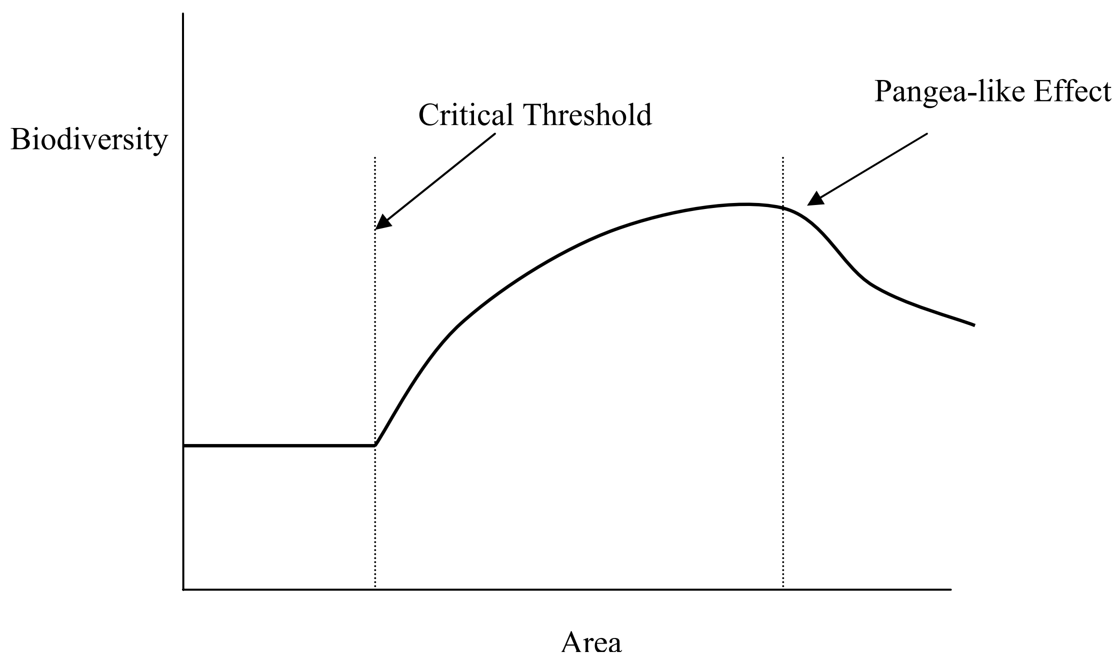

Figure 5

Conclusions

In this paper we argue that wetlands can be included as nutrient abating technology in a point-nonpoint source water quality trading program. However, we have shown that the program that does incorporate wetlands needs to take into account the ancillary benefits created by wetlands. This can be done in one of two ways: (1) the ancillary benefits are included in the market price for expected loadings permit and the point/nonpoint source trading ratio is adjusted to account for the ancillary benefits, or (2) the producer of wetlands can sell the nutrient trading capacity of the wetland in the nutrient market and the ancillary benefits are sold in another market, should one exist. We have shown a novel approach toward the adjustment of the point/nonpoint source trading ratio in that the choice depends on whether the wetland serves to reduce or increase the variance of the loadings from the nonpoint source of nutrients and the size of the marginal benefits. The choice of using one market versus multiple markets depends on the shape of the curve representing the marginal ancillary benefits. If the curve is relatively steep the policy maker should allow the nonpoint source to trade the wetland ancillary benefits in a separate market. If the marginal benefits curve is relatively flat, the policy maker should allow the nonpoint sources extra credit, through the corrected trading ratio, in the single market. We have introduced the idea, however that there is not necessarily agreement on the shape of the benefits curve, and indeed the curve may be different for different kinds or locations of wetlands. Future research includes a multidisciplinary approach to this problem wherein the benefits curves for several wetlands are measured empirically.

REFERENCES

Austin, D., A. Krupnick, V. McConnell. 1997. "Efficiency and Political Economy of Pollution Control witth Ancillary Benefits: An Application to NOx Control in the Chesapeake Bay Airshed." Discussion Paper 97-34. Resources for the Future, Washington, DC.

Biller D. 2003. Harnessing markets for biodiversity: towards conservation and sustainable use. Organisation for Economic Co-operation and Development, Paris, France.

Byström, O. 1998. "The Nitrogen Abatement Cost in Wetlands." Ecological Economics. 26(3): 321-331.

Byström, O., H. Andersson and I. Gren. 2000. "Economic Criteria for Using Wetlands as Nitrogen Sinks Under Uncertainty." Ecological Economics. 35(1): 35-45.

Doering, Otto C., Francisco Diaz-Hermelo, Crystal Howard, Ralph Heimlich, Fred Hitzhusen, Richard Kazmierczak, John Lee, Larry Libby, Walter Milon, Tony Prato, and Marc Ribaudo (1999) NOAA COASTAL OCEAN PROGRAM, Decision Analysis Series No. 20, "Evaluation of the Economic Costs and Benefits of Methods for Reducing Nutrient Loads to the Gulf of Mexico," Topic 6 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico.

Environmental Law Institute. 2005. National Forum on Synergies Between Water Quality Trading and Wetland Mitigation Banking. ELI Project No. 0508-01. Washington, DC.

Feng, H. and C. Kling. 2005. "The Consequences of Cobenefits for the Efficient Design of Carbon Sequestration Programs." Canadian Journal of Agricultural Economics. 53: 461-476.

Godard O. 2001. Domestic transferable permits for environmental management: design and implementation. Organisation for Economic Co-operation and Development, Paris, France.

Heal G. 2000. Nature and the marketplace: capturing the value of ecosystem services. Island Press, Washington, DC.

Heberling, M., H. Thurston, M. Mikota. 2007. "Incorporating Wetlands into Water Quality Trading: Economic Considerations." National Wetlands Newsletter. 29(1): Forthcoming.

Helfand, G. E. 1991. "Standards versus Standards: The Effects of Different Pollution Restrictions." American Economic Review. 81 (3): 622-634.

Horan, R. and J. Shortle. 2005. "When Two Wrongs Make a Right: Second-Best Point-Nonpoint Trading Ratios." American Journal of Agricultural Economics. 87(2):340-352.

Kieser and Associates. 2004. Ecosystem Multiple Markets. White Paper on behalf of The Environmental Trading Network. Accessed on March 21, 2006 at: http://www.envtn.org/docs/EMM_WHITE_PAPERApril04.pdf.

King, D., P. Kuch. 2003. “Will Nutrient Credit Trading Ever Work? An Assessment of Supply and Demand Problems and Institutional Obstacles.” Environmental Law Reporter. Environmental Law Institute, Washington, DC.

Malik, A., D. Letson, S. Crutchfield. 1993. “Point/Nonpoint Source Trading of Pollution Abatement: Choosing the Right Trading Ratio.” American Journal of Agricultural Economics. 75: 959-967.

Mitsch, William J. and James G. Gosselink (2000) Wetlands, 3rd edition, John Wiley and Sons, Inc., New York

Montero, J. 2001. “Multipollutant Markets.” The RAND Journal of Economics. 32(4): 762-774.

Moustafa, M. Z., M.J. Chimney, T.D. Fontaine, G. Shih, and S. Davis (1996) “The response of a freshwater wetland to long-term "low level" nutrient loads-Marsh Efficiency.” Ecological Engineering, 7, pp. 15-33.

Raffini, E. and M. Robertson. 2005. “Water Quality Trading: What Can We Learn from 10 Years of Wetland Mitigation Banking?” National Wetlands Newsletter, 27(4):July-August.

Ribaudo, M., R. Heimlich, R. Claassen, M. Peters. 2001. “Least-cost management of nonpoint source pollution: source reduction versus interception strategies for controlling nitrogen loss in the Mississippi Basin.” Ecological Economics. 37(2): 183-197.

Shortle, J. 1987. “Allocative Implications of Comparisons Between the Marginal Costs of Point and Nonpoint Source Pollution Abatement.” NJARE. 16: 17-23.

Spieles, D. J., and W. J. Mitsch (2000) “The Effects of Season and Hydrologic and Chemical Loading on Nitrate retention in Constructed Wetlands: A comparison of low and high-nutrient wetlands.” Ecological Engineering, 14, pp. 77-91.

Weitzman, M. 1974. “Prices vs. Quantities.” The Review of Economic Studies. 41(4): 477-491.

Woodward, R. and M. Han. 2004. Double Dipping in Pollution Markets. Paper presented at the Annual Meeting of the American Agricultural Economics Association. Accessed on March 21, 2006 at: http://agecon2.tamu.edu/people/faculty/woodward-richard/paps/DoubleDipping.PDF.

Progress Report on

Designing Incentives for Private Maintenance and Restoration of Coastal Wetlands

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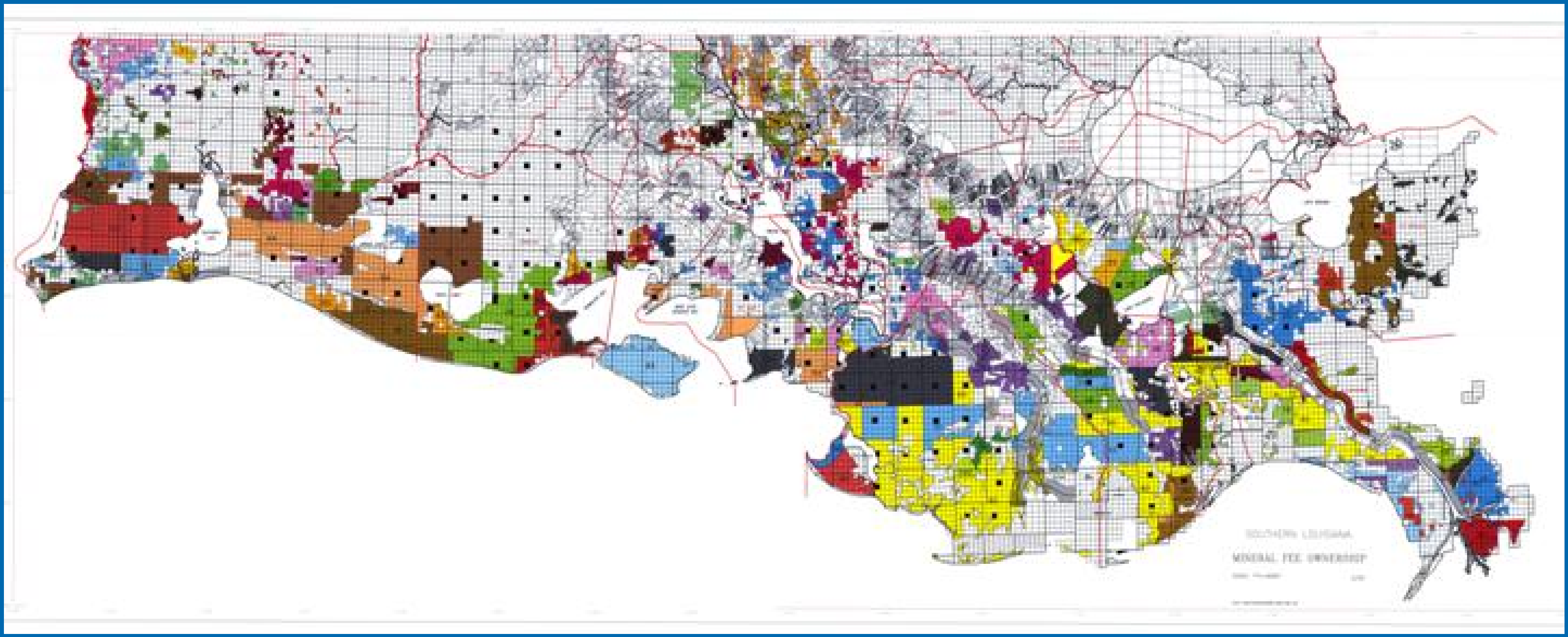
Landholdings in the Coastal Zone



- Coastal zone is primarily held privately
- Large tracts held by land / oil / gas companies
- Numerous small landholders throughout the coast

Threats To Louisiana's Coastal Wetlands

- coastal zone made up of low marshes and swamps susceptible to

  → subsidence / sea-level rise
  → erosion / oil & gas activity
  → vegetative destruction by nutria

- Channeling of rivers has directed alluvial sediments offshore

- Historical susceptibility to hurricanes – Katrina / Rita eliminated >200 sq. miles of wetlands, dispersed many landholders

Intervention to Meet the Threats

- Public Actions:
  → large-scale diversions, but operation at a fraction of capacity due to user conflicts
  → re-vegetation projects on a small-scale
  → land-rights for projects difficult to obtain

- Can Private Actions Help?
  → federal laws and programs recognize importance of local coastal stewardship
  → long history of water / land management in coastal agriculture
  → investment incentives needed to overcome increasing uncertainty and complexity

Project Goals

- Develop a framework for investigating incentive structures for private coastal management
- Determine attitudes and reactions to various incentives for private coastal management
- Examine how the combined socioeconomic and physiographic characteristics of landholdings affect private investment decisions
- Assess the efficacy of potential policy instrument designs aimed at private restoration activities

Current Progress

While participating in coastal triage and waiting for resettlement / locating of landholders:

- Estimated a hedonic model of coastal land value to determine the changes in private wetland values that would be forthcoming from various restoration scenarios
- Examined the ability of price-based incentives to encourage private maintenance of wetlands through nutria control programs

Example of Primary Data

am 29

LIMITED WARRANTY DEED

FILE#260681

STATE OF LOUISIANA §

PARISH OF CAMERON §
§

Conveyance

'99 JUN 23 PM 3 20

BE IT KNOWN, That on the date(s) shown below, before us the undersigned Notaries Public in and for the Counties or Parishes and States indicated below, duly commissioned and qualified, and in the presence of the undersigned, competent witnesses:

PERSONALLY CAME AND APPEARED: **AMOCO PRODUCTION COMPANY**, a Delaware corporation ("**Grantor**"), appearing by and through Kenneth Wayne Sweeney, its Attorney-in-Fact, and **JAMES C. ROBINSON**, 4315 Ashland Drive, Lake Charles, Louisiana 70605 ("**Grantee**"), who declare that, for and in consideration of a sum in excess of $ 300,000 cash in hand paid, the receipt and sufficiency of which are hereby acknowledged, Grantor does hereby GRANT, SELL, CONVEY, ASSIGN and DELIVER unto Grantee the real property situated in Cameron Parish, Louisiana, described in **Exhibit A** attached hereto and made a part hereof (the "Property") subject to zoning laws, regulations and ordinances of municipal and other governmental authorities, and any matters which are of public record affecting the Property and any encumbrances (to the extent they are in effect) described in **Exhibit B** attached hereto and made a part hereof (collectively, the "Permitted Encumbrances"), and subject to all of the provisions hereof.

TO HAVE AND TO HOLD the Property, together with all and singular the rights and appurtenances thereto in anywise belonging, unto Grantee, Grantee's successors and assigns forever, and Grantor does hereby bind itself and its successors and assigns to warrant and forever defend the title to the Property, as to Grantee, Grantee's successors and assigns against every person whomsoever lawfully claiming, or to claim the same, or any part thereof by, through or under Grantor, but not otherwise, subject, however, to the Permitted Encumbrances and the following provisions.

SAVE AND EXCEPT, and there is hereby reserved unto Grantor, its successors and assigns, the following:

(1) all rights in and to all oil, gas and other minerals, of every kind, both similar and dissimilar, in, on and under and that may be produced from the Property, together with the right of ingress and egress at all times for the purpose of mining, drilling, exploring, operating and developing the Property for minerals and removing the same therefrom, as further elaborated in Exhibit "B" item #3; and

(2) a servitude, easement and right-of-way for ingress, egress, and passage, at all times, over and across the Property to and from other lands currently owned by Grantor, or previously owned by Grantor and now owned by others, in Township 12 South, Ranges 12 and 13 West, Louisiana Meridian, Cameron Parish, Louisiana, provided, however, this servitude for ingress, egress, and passage shall be confined to existing roads, levees, trails, water channels, canals, pirogue trails and ditches, and such other passages as Grantee may reasonably designate from time to time for common usage.

BY THE ACCEPTANCE OF THIS DEED, GRANTEE TAKES THE PROPERTY "AS IS, WHERE IS" EXCEPT FOR THE WARRANTIES OF TITLE AS PROVIDED AND LIMITED HEREIN. GRANTOR HAS NOT MADE AND DOES NOT MAKE ANY REPRESENTATIONS AS TO THE PHYSICAL OR ENVIRONMENTAL CONDITION, LAYOUT, FOOTAGE, EXPENSES, ZONING, OPERATION, OR ANY OTHER MATTER AFFECTING OR RELATING TO THE PROPERTY, AND GRANTEE HEREBY EXPRESSLY ACKNOWLEDGES THAT NO SUCH REPRESENTATIONS HAVE BEEN MADE AND RELEASES GRANTOR FROM ANY AND ALL LIABILITY OF EVERY KIND AND CHARACTER WITH RESPECT THERETO, WHETHER OR NOT CAUSED BY OR ATTRIBUTABLE TO GRANTOR'S NEGLIGENCE. GRANTOR MAKES NO WARRANTIES, EXPRESS OR IMPLIED, OF MERCHANTABILITY, MARKETABILITY, FITNESS OR SUITABILITY FOR A PARTICULAR PURPOSE OR OTHERWISE. ANY IMPLIED WARRANTIES ARE EXPRESSLY DISCLAIMED AND EXCLUDED. THIS DEED IS EXPRESSLY SUBJECT TO THE TERMS AND CONDITIONS OF THAT CERTAIN AGREEMENT

1

EXHIBIT "A"

<u>TRACT 1:</u>

Approximately 2300 acres of land located in <u>unofficial</u> Sections 1, 11-14 and 22-27, T12S-R13W, Louisiana Meridian, Cameron Parish, Louisiana (based on that certain <u>unadjudicated</u> Agreement Survey of a portion of said township made by A. B. Hill for Pan American Petroleum Corp. (predecessor of Amoco), survey plat for which is dated July 14, 1969, and filed in-house in Amoco's Land Survey Engineering group located in Houston, Texas), being a portion of the same property acquired by Stanolind Oil and Gas Company (predecessor of Amoco) from Wright Morrow by Act of Sale dated July 31, 1935, recorded in COB 27 at Page 280 of the Conveyance Records of said Parish, said land being more particularly described as follows:

All of the captioned <u>Section 1</u>.

The following portions of the captioned <u>Section 11</u>: The East half (E/2) of the Southeast quarter (SE/4) of the Northeast quarter (NE/4) of the Southwest quarter (SW/4) of the Northwest quarter (NW/4); the East half (E/2) of the East half (E/2) of the Southeast quarter (SE/4) of the Southwest quarter (SW/4) of the Northwest quarter (NW/4); the Southeast quarter (SE/4) of the Northwest quarter (NW/4); the North half (N/2) of the North half (N/2) the Northeast quarter (NE/4) of the Northeast quarter (NE/4) of the Southwest quarter (SW/4); the Southeast quarter (SE/4) of the Northeast quarter (NE/4) of the Northeast quarter (NE/4) of the Northeast quarter (NE/4) of the Southwest quarter (SW/4); the East half (E/2) of the Southeast quarter (SE/4) of the Northeast quarter (NE/4) of the Northeast quarter (NE/4) of the Southwest quarter (SW/4); the Northwest quarter (NW/4) of the Southeast quarter (SE/4); all that portion of the East half (E/2) of the Southwest quarter (SW/4) of the Southeast quarter (SE/4), the West half (W/2) of the Southwest quarter (SW/4) of the Southeast quarter (SE/4), and the Southeast quarter (SE/4) of the Southeast quarter (SE/4) of the Southeast quarter (SE/4) of the Southwest quarter (SW/4) lying southeasterly of a line running five feet (5') northwesterly of, and parallel to, the centerline of that certain existing levee which traverses northeasterly-southerwesterly through said aliquot parts; and all that portion of the North half (N/2) of the Northeast quarter (NE/4) of the Northwest quarter (NW/4) of the Southwest quarter (SW/4) of the Southeast quarter (SE/4), and the North half (N/2) of the North half (N/2) of the Northeast quarter (NE/4) of the Southwest quarter (SW/4) of the Southeast quarter (SE/4) lying northwesterly of a line running five feet (5') northwesterly of, and parallel to, the centerline of the above said levee.

Hedonic Model of Coastal Property Value

ln(PRICE) = ß0 + ß1*ln(ACRES) + ß2*ln(FRESH/OW) +
ß3*ln(INTER/OW) + ß5*ln(BRACKISH/OW) +
ß6*ln(OTHER/OW) + ß7*ln(DISTANCE) +
ß8*ln(DROAD) + ß9*ln(SEPARATE)

Variable	Estimate	S.E.
Intercept	5.3220*	0.4774
ln(acres)	-0.0179	0.0459
ln(fresh/ow)	0.0403*	0.0134
ln(inter/ow)	-0.0573*	0.0147
ln(brack/ow)	-0.0203	0.0123
ln(other/ow)	0.0263*	0.0129
ln(dist)	0.2588*	0.1351
ln(road)	-0.0643	0.0463
ln(separate)	-0.2290	0.1440

Preliminary Implications

- The type of wetland present affects property value in different ways:

  positive effect – freshwater marsh, non-marsh

  negative effect – intermediate marsh

- As distance from the coast increases, property value increases

- Estimated price differentials suggests that incentive programs may need to be tailored around wetland types

Ongoing Work

- Expanding the dataset to include coastal parishes in the central and southeastern parts of the state

- Will include analysis of historical wetland loss (1960-2000) on property value

- Will include analysis of 'expected' wetland loss (possibly to 2050) on property value

Impact of Bounties on Nutria Harvests

Average Cost Model

$$\ln(H)_t = \beta_0 + \beta_1 \cdot \ln P_t + \beta_2 \cdot P_t + \beta_3 \cdot OC_t + \beta_4 \cdot alligator_t + \beta_5 \cdot freeze_t + \beta_6 \cdot cncp$$

Parameter	Estimate	S.E.	P-value
intercept	-3.9594	0.3640	<0.0001
ln(price)	2.5828	0.2974	<0.0001
price	-0.1792	0.0386	<0.0001
opportunity cost	0.1129	0.0242	<0.0001
alligator	-0.0174	0.0060	0.0064
freeze	0.0025	0.0025	0.3247
cncp	0.5584	0.2654	0.0422
DW = 1.80	SSE = 3.0191	MSE = 0.0816	

Impact of Bounties on Nutria Harvests

Marginal Cost Model

$$\ln(H)_t = \beta_0 + \beta_1 \cdot \frac{1}{P_t} + \beta_2 \cdot OC_t + \beta_3 \cdot alligator_t + \beta_4 \cdot freeze_t + \beta_5 \cdot cncp$$

Parameter	Estimate	S.E.	P-value
intercept	0.7606	0.1246	<0.0001
1/price	-6.8830	0.5171	<0.0001
opportunity cost	0.1014	0.0208	<0.0001
alligator	-0.0180	0.0056	0.0027
freeze	0.0038	0.0026	0.1448
cncp	0.3231	0.2393	0.1850
DW = 1.81	SSE = 2.9625	MSE = 0.0779	

Preliminary Implications

- Data described by either the common or private property model – rights regime is mixed
- Average cost model can be used to predict harvests associated with different bounty levels

Bounty Level ($/tail)	Estimated Harvest (#)
4	253,000
6	425,588
8	651,574
10	831,477
12	956,279
14	1,022,497

Next Steps

- Finish pre-testing questionnaire
- Collection of field data from large (personal interview) and small (mail, telephone survey) landholders
- Estimate a double-hurdle Tobit model of restoration investment decision making
- Combine the analyses to assess the role of existing / potential policy instruments for encouraging restoration

Marc Ribaudo Comments for Market Mechanisms Workshop

An Experimental Exploration of Voluntary Mechanisms to Reduce Non-Point Source Water Pollution With a Background Threat of Regulation – Suter, Segerson, Vossler, and Poe.

In this paper an ambient-based tax policy as a regulatory back-up to voluntary adoption of management practices. The issue is to design the back-up to maximize the incentive for voluntary action. The goal is to find most efficient policy design.

The voluntary/regulatory policy is a very good subject for research, as this is the framework for addressing NPS. Section 319 of the Clean Water Act requires each State to: (1) identify navigable waters that, without additional action to control nonpoint sources of pollution, cannot reasonably be expected to attain or maintain applicable water-quality standards or goals, (2) identify nonpoint sources that add significant amounts of pollution to affected water, and (3) develop a NPS management plan on a watershed basis to control and reduce specific nonpoint sources of pollution. Among other things, the management plan is required to contain a list of best management practices (BMPs) for controlling NPS pollution, a timetable for implementing the plan, and enforceable measures to ensure the plan is implemented. Implies some sort of back-up regulations.

There are two basic problems with ambient based taxes that may severely limit what can be accomplished in practice. First, each producer must have some expectation of how his/her actions affect ambient measure. This means knowledge of fate and transport. Second, for this policy to work it requires that each firm or producer has some knowledge or expectation of how each other landowner in the watershed behaves. This is borne out by results reported in the paper, where allowing conversation between participants in the economic experiment resulted in more efficient results. Assuming this is the case, the transactions costs of such communications could be quite high, depending on the size of the watershed. Transactions costs are not accounted for in this research. Could there be a role for a central clearinghouse? Would producers be willing to reveal private information for the good of the regulated community?

This line of research has invariably used a tax as the regulatory back-up. The paper indicates that such a policy design can work. In reality, environmental taxes are taboo in this country, and it is not likely that this will change any time soon.

So the question is: can an efficient (or relatively efficient) program be developed that allows voluntary compliance with a regulatory back-up based on input or technology standards? Several States use triggers that result in regulations. Nebraska protects groundwater from nutrients with a policy whereby N concentrations trigger different nutrient management requirements. California uses a similar approach to protect groundwater from pesticides. Vermont allows voluntary adoption with financial assistance of recommended BMPs, but will require BMPs if progress towards water quality goals is not made. .

Conservation compliance is mentioned in the paper as footnote. The penalty based on input decisions rather than on actual effluent generated. This is a second best solution, but it is practical and it apparently works.

A more promising line of research from a policy perspective might be to examine the design of a program that uses the threat of technology-based standards to provide the appropriate incentives to spur voluntary action. Mandatory practices would provide less flexibility than voluntary actions. Adding a flat penalty could provide an additional incentive to act "voluntarily" Does this approach provide an adequate incentive to act voluntarily? I have not seen much on this.

Choice Experiments to Assess Farmers' Willingness to Participate in Water Quality Trading Market – Peterson, Fox, Leatherman, and Smith

This paper looked at factors behinds farmers' willingness to participate in water quality trading programs. Lack of participation in trading programs (supply-side impediment) is a real issue. This is particularly true given the joint USDA-EPA announcement in support of water quality trading projects across the country.

USDA not only emphasizing markets for water quality, but other environmental markets as well, such as carbon sequestration, wetland mitigation, and wildlife habitat (hunting). Such markets are seen as a means of increasing conservation through private funding. Increasing our knowledge about farmer willingness to participate in such markets is critical to the development and success of markets where farmers make a major contribution. We have much to learn about participation, so there is a danger of rushing ahead and being disappointed.

Research reported in paper not yet complete, so no results as of yet.

In the case of WQT, it has been suggested that being associated with regulatory programs (cap and trade on point sources) is a deterrent. Farmers are afraid that by selling abatement credits they are admitting they are polluting, and that they would be the next targets for regulation. This particular issue is not covered in this paper, but needs additional examination. This has an important bearing on the issue, since cap and trade is the best way to create demand for services from agriculture. This issue may only be a concern for pollutants created by agriculture. Agriculture is widely recognized as a net sink for carbon, so there would be no "stigma" attached to participating in a market for carbons. Not so with water quality, where agriculture is also contributes pollution.

Incorporating Wetlands in Trading Programs: Economic and Ecological Considerations – Thurston and Heberling

This paper looks at incorporating wetlands into water quality trading programs. The issue is wetlands also produce ancillary benefits, so they should be encouraged over other nutrient-reduction strategies. This would also increase the number of participants in trading markets. “Thin” markets are seen as one of the issues raised by King and Kuch as preventing markets from operating efficiently. Allowing wetland restoration would conceivably increase the number of credit suppliers.

There are several issues here. First is the notion that lack of supply is a major stumbling block for trading. Increasing the size of the market by allowing wetlands will increase supply only. In many watersheds with nonpoint sources, there are already more sellers than buyers. Also, one of the few examples of a point-nonpoint trade involved a single buyer and four farms. This is not trading in the classical sense, but an offset. However, society still benefited. Having few participants does not necessarily prevent beneficial trades from occurring.

Are created wetlands more likely to participate in trading markets? If the creators are the same individuals not willing to participate now, what is gained?

Another issue is the potential competition for supply-side credits. Wetlands filter runoff from upstream. If upstream farmers agree to participate in the market, install BMPs and sell credits, the utility of wetland declines. The supply of credits from agriculture is not strictly additive. Potential interactions with neighbors’ decision need to be taken into account. The transactions costs for estimating credits and developing side contracts could be high.

The paper present 2 approaches for incorporating wetlands:

- Capture ancillary benefits in the nutrient trading market by adjusting trading ratio, and therefore credit prices
- Separate markets for ancillary benefits

The latter seems to be the most palatable. One of the requirements of a successful trading market is that point and nonpoint sources produce the same good. In the first approach, a point source purchaser is looking for abatement credits for a particular pollutant, say nitrogen. However, a wetland creator is selling something that is different: nitrogen abatement PLUS some other ancillary benefit. If the credit price is lower, and point sources needs are met, then everything fine. But the results presented in the paper show that prices could increase. This would put credits from wetlands at a competitive disadvantage to simple nutrient management.

It seems the simplest approach would be to market services separately. Now, in many cases, markets won’t exist for the simple reason that environmental services take on the characteristics of public goods. The traditional way of handling this is for government or land trusts to purchase benefits for society at large. Targeting wetland creation through public or other programs in watersheds with trading programs could be a way of rewarding wetland creation. It is ironic that the Wetland Reserve Program is the only conservation program that specifically prohibits the owner of created wetlands from selling environmental credits created by the restoration.

Shortle Discussion

Introductory Comments

- I have thoroughly enjoyed this conference. The papers that have been presented over the two days have been consistently interesting. Congratulations to EPA for the quality of work it has funded and the other participants it has invited.

- My comments about the quality of the papers at the workshop apply equally to those in this session.

- Prior sessions explore areas in which we have had much more experience with market mechanisms. Water is a new frontier that poses a lot challenges. These papers will help address those challenges.

An Experimental Exploration of Voluntary Mechanisms to Reduce Non-Point Source Water Pollution With a Background Threat of Regulation

Jordan Suter, Cornell University, Kathleen Segerson, University of Connecticut, Christian Vossler, University of Tennessee, and Greg Poe, Cornell University

- This paper comes from an interesting research program – ambient based instruments have received lots of attention in theory but not much in practice. They rely on very strong but assumptions about equilibrium behavior. Thus experimental testing of the type done in this research is clearly the way to go to learn how they might work in practice.

- The specific application examines an ambient tax as a threat to induce voluntary adoption of nonpoint pollution control practices. This too is interesting.

- Now I like the paper a lot, and it is distinctly a contribution to the economics literature on ambient instruments. But, I don't see it as squarely addressing the nonpoint problem. Some missing elements:

    - Observability of emissions – emissions are assumed unobservable by the environmental prinicpal, but observable and deterministically controllable by the agents.

    - Perfect mixing – no spatial heterogeneity or uncertainty

    - Stochasticity

    - Multiple choices, reliability and complexity

    - Different types – large versus small –

    - Capital and adjustment costs

- Policy Relevance?

    - The NPS economics literature versus the NPS policy problem

    - Group penalties are unlikely to happen –

        - partly because they are politically nonstarters,
        - partly because patently violate common sense notions of fairness
        - and there are better alternatives

Incorporating Wetlands in Water Quality Trading Programs: Economic and Ecological Considerations

Hale Thurston and Matthew Heberling, EPA, National Risk Management Research Laboratory, Cincinnati, Ohio

- Hale & Hebberling paper more directly embraces the complexity of nps pollution and a policy approach that is of greater interest
    - Outcomes a function of practices
    - Stochastic
    - Trading
- But most importantly – the complexity of the externalities that result from land use practices
    - In this case wetlands
        - Water quality
        - Carbon sequestration
        - Habitat service
- More generally,
    - Open space amenities
    - Air pollution
- These are joint products of production choices: The challenge – how to design policy instruments to address the set of outcomes?
- Currently, ag policies are highly uncoordinated, and often conflicting.
- Separate policies or multi-purpose?
- Theory of policy would suggest a tool for each target, but if not, then how to modify those we can take up.
- Trade ratio in "exchange type market." The next step would be to look at the levels of nps permits
- But more promising may be to reconsider the type of market – contracts for explicit services – this is where I am going in model reliability

Choice Experiments to Assess Farmers' Willingness to Participate in a Water Quality Trading Market

Jeff Peterson, Washington State University, and Sean Fox, John Leatherman, and Craig Smith, Kansas State University

- Non-participation is a huge issue in water quality markets – this paper explores how to increase participation by design "friendlier" markets

  Reasons for noparticipation:

    No gains from trade

      No cost heterogeneity? ?

      No cap?

      Bad rules?

    Coordination failures

      Mechanisms for bringing buyers and sellers together

    Implicitly, accepting liability for water quality problems

- Interesting approach – a few issues

  How were the attributes for the choice experiments selected?

  What are the policy implications of results on attributes?

  Lack of context about coordination mechanism

  Sample not random