# The Social Cost of Trading: Measuring the Increased Damages from Sulfur Dioxide Trading in the United States

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## *Abstract*

*The sulfur dioxide ($SO_2$) cap and trade program established in the 1990 Clean Air Act Amendments is celebrated for reducing abatement costs ($0.7 to $2.1 billion per year) by allowing emissions allowances to be traded. Unfortunately, places with high marginal costs also tend to have high marginal damages. Ton-for-ton trading reduces emissions in low damage areas (rural) while increasing emissions in high damage areas (cities). From 2000 to 2007, conservative estimates of the value of mortality risk suggest that trades increased damages from $0.8 to $1.1 billion annually relative to the initial allowance allocation and from $1.5 to $1.9 billion annually relative to a uniform performance standard. With U.S. Environmental Protection Agency (USEPA) values, trades increased damages from $2.4 to $3.2 billion annually compared to the initial allowance allocation and from $4.4 to $5.4 billion compared to a uniform performance standard. It is not clear that the ton-for-ton $SO_2$ cap and trade program is actually more efficient than comparable command and control programs. The trading program needs to be modified so that tons are weighted by their marginal damage.* 

## INTRODUCTION

Since the original essay on emissions allowance trading (Dales, 1968), economists have been very excited about the reductions in abatement costs that trading encourages (Tietenberg, 1980). However, some pollutants cause different degrees of damage depending on where they are emitted. In such contexts, scholars have recognized that, by influencing the location of emissions, trading may change the total severity of damages. For example, trading could cause emissions to migrate from low marginal damage to high marginal damage locations and increase total damage. One solution is to limit markets so that trading could only occur between sources whose marginal damages are similar (Tietenberg, 1980). However, this creates thin markets that have other problems. The Acid Rain Program (ARP) established by the 1990 Clean Air Act Amendments (CAAA) consequently favored a national market for allowance trades to encourage more participants and more trading (Nordhaus, 2000).

Trading programs improve cost effectiveness by allowing high marginal abatement cost firms to trade with low marginal abatement cost firms. Initial estimates for the sulfur dioxide ($SO_2$) cap and trade program during the limited Phase I

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period between 1995 and 1999 suggest abatement cost savings of \$150 to \$400 million annually (Schmalensee et al., 1998; Keohane, 2006). Phase II abatement cost savings were even larger because even more facilities were included. The estimated abatement cost savings from trading in the Phase II period between 2000 and 2009 range from \$0.7 to \$2.1 billion per year (Carlson et al., 2000; Keohane, 2006; Ellerman et al., 2000).

Although several ARP scholars recognized that trading implies emissions are changing location, they argued that trading had no important consequence on damages (Burtraw et al., 1998; Burtraw & Mansur, 1999; Shadbegian, Gray, & Morgan, 2007). We argue these analyses were too limited. First, some of the analyses only examined abatement cost savings from trades (Carlson et al., 2000). Other studies focused on whether the trades would lead to hot spots—places with high concentrations of emissions (Hausker, 1992). This did not occur because the ARP discouraged trades into areas that did not meet $SO_2$ National Ambient Air Quality Standards (NAAQS) (Hausker, 1992). Finally, some analyses examined the effects of trades on health benefits but relied only on state-level data that obscured the effect of trades between rural and urban locations (Burtraw et al., 1998; Burtraw & Mansur, 1999; Shadbegian, Gray, & Morgan, 2007). The previous literature consequently failed to detect any effect on damages caused by trading.

In this paper, we argue that the current approach to $SO_2$ trading caused a substantial increase in damages. We argue that trading encouraged emissions to move from rural (low damage) to urban (high damage) areas. Of course, it was not the intent of the ARP to increase damages. The purpose of trading was to move abatement from high marginal cost to low marginal cost firms to secure abatement cost savings. If damages were randomly allocated across sites, there would be no net damage effect. However, marginal costs and marginal damages happen to be positively correlated for $SO_2$. Before the ART, firms in nonattainment areas that exceeded the NAAQS had more stringent emission rules and therefore higher marginal abatement costs. Urban areas often exceeded the NAAQS because cities emit half of all local air pollution even though they make up only 3 percent of total land area (Muller & Mendelsohn, 2009). The density of emissions leads to high ambient pollution concentrations. Hence, marginal abatement costs tend to be higher for firms in urban locations. Urban areas also have higher marginal costs because rents and wages are higher.

Sources in urban areas are also located near high population densities and consequently they have high marginal damages (Muller & Mendelsohn, 2009; Muller, Tong, & Mendelsohn, 2009). Because $SO_2$ trading encourages emissions to move from low marginal cost to high marginal cost locations, $SO_2$ trading also led emissions to go from low marginal damage (rural) to high marginal damage (urban) counties. The overall effect was to increase annual damages. This analysis explores just how much total damages increased because of $SO_2$ trading under the ARP.

This paper tracks the movement of $SO_2$ allowances from the initial allocation of permits to each firm to the location of the final emission after trading. This study focuses on Phase II of the ARP for the years 2000 through 2007 and covers almost all fossil fuel electric generating stations in the United States. We calculate the damages from actual emissions and compare them to the level of damages that would have occurred at the initial allowance allocation prior to trading. To test the robustness of our findings, we also compare damages from actual emissions to damages under a uniform emissions standard defined by heat input. By holding total emissions fixed, these simulations isolate the impact of the spatial reallocation of emissions due to trading.

The study relies on an integrated assessment model—the Air Pollution Emission Experiments and Policy analysis (APEEP) model (Muller & Mendelsohn, 2007, 2009; Muller, Mendelsohn, & Nordhaus, in press)—to account for damages to human health, visibility, crops, recreation, and timber. The model incorporates the

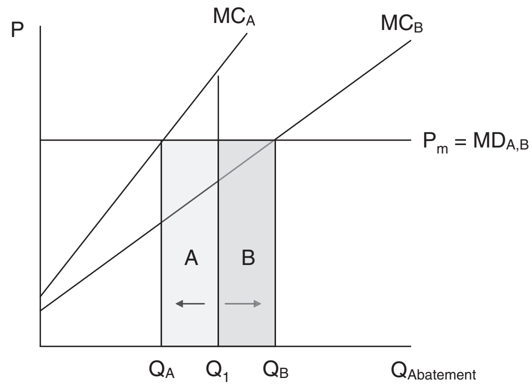



**Figure 1.** Effect of trading when marginal damages are equal across firms.

dispersion of emissions, the exposure of sensitive populations (especially humans), the response to the increased exposure, and then the sum of the values of the resulting damages. The integrated assessment model is used to compute marginal damages for emissions in each county in the U.S. and for over 600 individual point sources (Muller & Mendelsohn, 2009).

These marginal damages are then used to compute the change in damages associated with the $SO_2$ cap and trade program by comparing the damages from emissions both before and after the trades. While the total amount of emissions remains fixed in these two scenarios, the location of emissions changes according to whether or not trading is permitted. Employing APEEP default assumptions for the valuation of mortality risks, damages are estimated to have increased by \$0.8 to \$1.9 billion per year from 2000 to 2007. Using the higher valuation of mortality risk that is employed by the U.S. Environmental Protection Agency (USEPA), damages are estimated to have increased by \$2.4 to \$5.4 billion per year. Depending on the values used, the increased damages are at least as large if not larger than the abatement cost savings from emission trading.

## CONCEPTUAL MODEL

Earlier studies have recognized that a positive (negative) covariance between marginal costs and marginal benefits of abatement can cause allowance trading on a ton-for-ton basis to increase (decrease) damages, relative to a command and control policy (Mendelsohn, 1986). However, the importance of the problem depends on the heterogeneity of marginal damages and the magnitude of the correlation. For example, if the marginal damages are almost the same everywhere, the effect of trading on aggregate damages will be minimal. In Figure 1 we assume that the marginal damages are alike for each firm and the price for pollution allowance ($P_m$) is also equal to marginal damages. Firm A is assumed to have high marginal costs of abatement and Firm B has low marginal costs. If both firms begin by executing the same level of abatement ($Q_1$), they both have an incentive to trade allowances; Firm A would like to buy ($Q_1 - Q_A$) allowances because at the prevailing price ($P_m$), buying allowances is cheaper than abatement. Conversely, Firm B has an incentive to sell ($Q_B - Q_1$) allowances because the resulting earnings exceed the additional abatement costs. There is an incentive to trade permits until $MC_A = MC_B = P_m$. These trades would reduce aggregate abatement costs. Aggregate damages remain unchanged because the aggregate increased damages by Firm A (rectangle A) is the

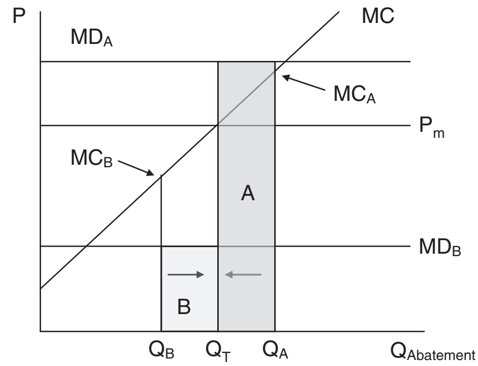



**Figure 2.** Effect of trades between urban and rural sources.

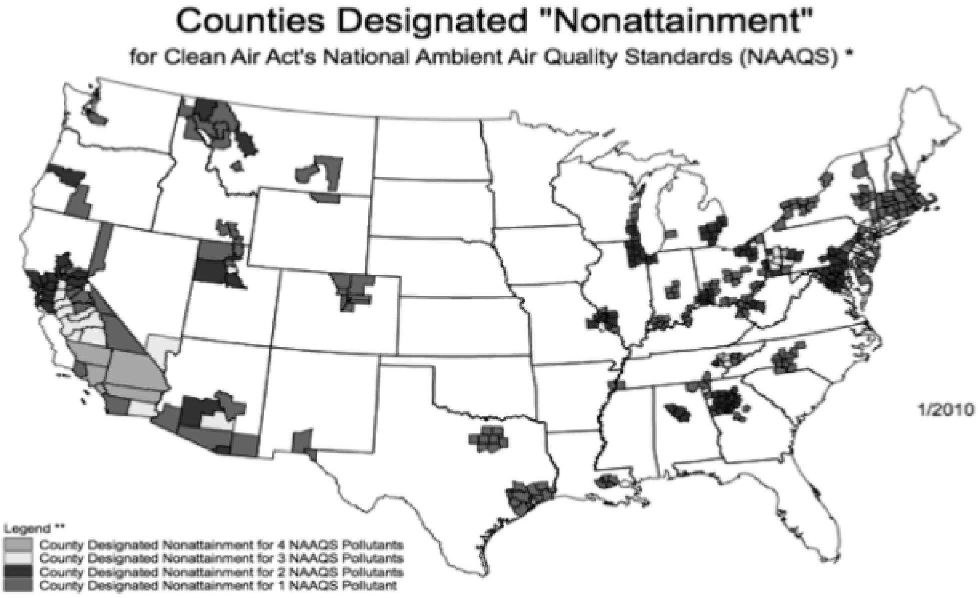
same as the aggregate reduced damages (rectangle B) by Firm B. This is the classic argument on behalf of trading allowances on a ton-for-ton basis. The classic assumptions clearly apply to some pollutants, notably greenhouse gases, where marginal damages do not depend on location. They also apply when the marginal cost of abatement is uncorrelated with the marginal damage at each location.

However, for many local pollutants the location of emission determines the level of marginal damages (Muller & Mendelsohn, 2009). Emissions that occur near sensitive populations lead to higher damages. If marginal costs are correlated with marginal damages, the spatial reallocation of emissions due to trading will have an effect on aggregate damages (Mendelsohn, 1986). A positive correlation suggests trading would increase damages and a negative correlation suggests trading would decrease damages.

Figure 2 illustrates this point using an urban firm (Firm A) and a rural firm (Firm B). Although the marginal abatement cost function of both firms could be identical, the NAAQS regulations force Firm A in the urban area to abate more than Firm B in the rural area ($Q_A > Q_B$). Firm A has higher marginal abatement costs than Firm B ($MC_A > MC_B$). The marginal damages are also not the same for the two firms. Emissions from Firm A cause higher marginal damages than those from Firm B because they lead to more human exposures, and health is the primary damage of $SO_2$ ($MD_A > MD_B$). We draw the marginal damage functions as horizontal lines because a single firm has only minute impacts on ambient concentrations and so its emissions alone do not alter marginal damages (Muller & Mendelsohn, 2009).

Because $MC_A$ is initially greater than $P_m$, the urban firm will want to buy permits from the rural firm so that it can abate less. Similarly, the rural firm will want to sell permits and pollute less because the $MC_B$ is less than $P_m$. The trading will reduce aggregate abatement costs. However, because the trading will encourage emissions to move from the rural to the urban area, damages will increase. The increase in damage (rectangle A) due to Firm A emitting more is larger than the reduced damage (rectangle B) due to Firm B abating more. The magnitude of the cost savings and damage increase is an empirical question. The costs savings have been explored in the literature. This paper quantifies the increase in damages.

Ton-for-ton trading programs are designed to reduce total abatement costs. The relative magnitude of the change in damages depends on the degree of heterogeneity in the marginal damages across firms relative to the variation in marginal costs and the relative slopes of the marginal cost and marginal damage functions (Mendelsohn, 1986).




*Source:* USEPA (2010b).

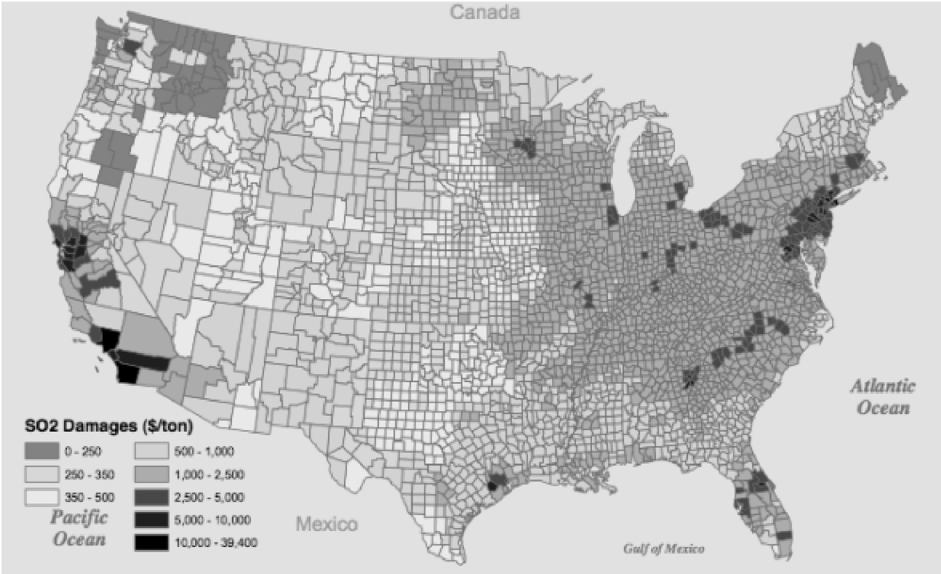
**Figure 3.** County level National Ambient Air Quality Standards nonattainment status.

Given the earlier reliance on NAAQS regulations, the positive correlation between marginal costs and marginal damages is likely to be a general phenomenon for many local pollutants. Urban areas will tend to have higher marginal damages per unit of emission for any damage related to people, whether it is direct impacts on health, enhanced depreciation of material possessions, or other nonmarket services such as visibility. Firms located in urban areas are also likely to face higher costs. First and foremost, emissions tend to be densely packed in urban areas, leading to higher concentrations. Concentrations in urban areas are likely to violate the NAAQS, forcing more stringent local abatement requirements. Second, the price of land and labor and even material inputs can be higher in urban areas than in rural areas. It therefore will often be the case that there is a positive correlation between marginal damages and marginal costs of abatement.

In Figure 3, we demonstrate which areas are in nonattainment with (violate) the NAAQS. As can be seen in the figure, most of the nonattainment areas are urban. Very few rural areas are in nonattainment. Consequently, urban areas face stricter emission control policies. Figure 4 reveals the marginal damage caused by emissions in different locations. It is clear that the marginal damages in urban areas are quite a bit higher than the marginal damages in rural areas. The positive spatial correlation by county in Figures 3 and 4 is striking. Note that a similar analysis at the state level would not reveal this correlation because the differences in costs and damages are driven by urban–rural land use, not state location.

## EMPIRICAL MODEL

To determine the damages caused by $SO_2$ emissions from the facilities governed by the ARP, we employ the Air Pollution Emission Experiments and Policy analysis model (APEEP) (Muller & Mendelsohn, 2007, 2009; Muller, Mendelsohn, & Nordhaus, in press). APEEP is a traditional integrated assessment model of air pollution for




*Source:* Muller and Mendelsohn (2009).

**Figure 4.** County-level $SO_2$ marginal damages.

the contiguous United States. APEEP is like other integrated assessment models that have been used by the USEPA to evaluate the benefits and costs of the Clean Air Act (USEPA, 1999). It follows a six-step process connecting the source of emissions, the air quality model, the resulting concentrations, the exposure of sensitive populations to concentrations, the dose response relationship between exposures and physical effects, and the valuation of these physical consequences in dollars (see Figure 5). It calculates damages for emissions of six air pollutants (sulfur dioxide, nitrogen oxides, volatile organic compounds, ammonia, fine particulate matter, $PM_{2.5}$, and coarse particulate matter, $PM_{10}$–$PM_{2.5}$). The effects encompassed by the model calculations are: adverse effects on human health, decreased timber and agriculture yields, reduced visibility, enhanced depreciation of materials, and reductions in recreation services. Schematically APEEP is represented in Figure 5.

County emissions for each criteria air pollutant (excluding carbon monoxide and lead but including ammonia) come from the USEPA's 2002 National Emission Inventory (USEPA, 2006). The air quality model is based on the Gaussian plume model (Turner, 1994). However, the model has been enhanced to capture chemical reactions. For example, the model captures the transformation of $SO_2$ into ammonium sulfate, a component of $PM_{2.5}$. The predicted pollution concentration levels by APEEP are comparable to predictions generated by the Community Multiscale Air Quality Model (Byun & Schere, 2006), which is considered the state-of-the-art air quality model (Muller & Mendelsohn, 2007).

Population-weighted exposures are computed by multiplying county-level populations times county-level pollution concentrations. The relevant populations for $SO_2$ and its derivatives include the number of people and the inventory of manmade materials for each county in the contiguous United States. Each type of exposure is computed separately. Exposures are translated into physical impacts through the use of dose response functions, which are drawn from the peer-reviewed literature. The relevant dose response functions in this example include premature mortalities, cases of illness, and enhanced depreciation of manmade materials. The literature

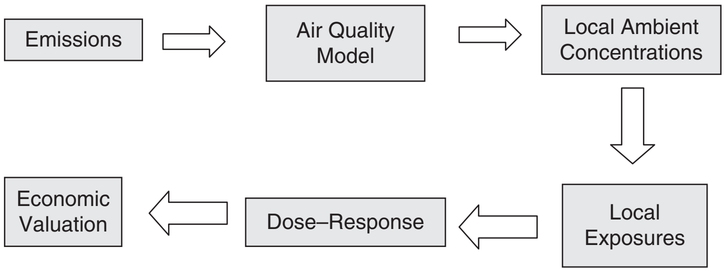



**Figure 5.** The APEEP model structure.

suggests that the single most important concentration–response relationship is that between (adult) human mortality and chronic exposures to $PM_{2.5}$ (USEPA, 1999; Muller & Mendelsohn, 2007, 2009). To model the mortality impacts of $SO_2$ emissions, APEEP uses the concentration–response relationship reported in Pope et al. (2002).

APEEP uses a valuation function to express these physical effects in monetary terms. For manmade materials, APEEP measures the effect of $SO_2$ exposure on painted surfaces, carbonate stone, and carbon steel. The damage to materials is the present value of the additional maintenance required to maintain the lifetime of the materials. For visibility, damages are valued using contingent valuation methods based on household willingness-to-pay for incremental changes in visibility associated with recreation experiences (Chestnut & Rowe, 1990). For acute health impacts, such as hospital admissions for asthma, APEEP uses the cost of illness for specific illnesses reported by USEPA (1999). However, the majority of the health costs are for mortality risks.

APEEP employs the results of hedonic wage studies to measure the monetary damage of small increases in mortality risk (see Viscusi & Aldy, 2003). However, rather than applying the same value to mortality risks faced by populations of all ages, APEEP adjusts the value to the remaining life years of the exposed populations. Remaining life years are tied to 19 age–sex cohorts. For each age cohort, APEEP computes the present value of the sum of life years remaining. The value of a life year is calculated from the wage premium and life years remaining of a 45-year-old male (which correspond to the mean age of the sample in hedonic wage studies). In these calculations, a 3 percent discount rate is used to calculate present values.

The mortality estimate of $2 million per value of statistical life (VSL), which is intended to reflect the value of a small risk, comes from a review of the hedonic wage literature (Mrozek & Taylor, 2002). In a sensitivity analysis, we also explore using $6 million per VSL, which corresponds to the USEPA's preferred value (USEPA, 1999). In the sensitivity analysis, we also apply the USEPA assumption that the VSL is the same across all ages. However, we do not employ the USEPA assumption that mortality impacts occur following a lag period after exposure.

We rely on estimates of the marginal damage of emissions in each county of the U.S. produced by APEEP (Muller & Mendelsohn, 2009). The marginal damage for $SO_2$ emissions is computed for each facility governed by the ARP. This modeling strategy captures the difference in damages per ton of $SO_2$ emissions between sources in urban versus rural counties.

A simple algorithm is used to calculate marginal damages. First, national baseline damages are estimated by APEEP from baseline emissions reported across the entire U.S. (USEPA, 2006). Next, APEEP adds 1 ton of $SO_2$ to baseline emissions at one regulated facility. APEEP then recalculates the total national damages. The change in total damages between the baseline and the run with the additional ton of $SO_2$ is the marginal damage of that emission. This experiment is repeated for each location covered in this study. Note that this approach captures the impact of secondary pollutants that result from the $SO_2$ emission.

We then estimate the impact of trading on damages. The allowance allocation and the actual emissions data for the original facilities governed by the ARP are collected from the USEPA Clean Air Markets Data (USEPA, 2010a). The ARP legislation mandated facilities receive their Phase II allowances at the initial allocation in 1993. There were 754 generating facilities that originally received allowances. These 754 facilities contain approximately 2,200 electric generating units (EGUs) and account for approximately 99.8 percent of the retired allowances (emissions) in Phase II of the program (2000 to 2009) (authors' calculations from USEPA, 2010a). The remaining 0.2 percent of retired allowances comes mostly from facilities that did not exist at the time of the initial allocation.

We begin the analysis by calculating total damages given observed $SO_2$ emissions. Actual emissions take into account traded allowances across firms. This estimate of damages based on observed emissions is compared to the damages associated with two counterfactual scenarios. In all cases, total emissions of $SO_2$ are identical.

In the first counterfactual scenario, we assume that firms would have equated emissions to the initial allowance allocation. The motivation is that the initial allocation likely reflected existing laws and regulations before trading began. In the second counterfactual, we assume a performance standard forces all plants to have the same $SO_2$ emission rate per heat input, measured in British thermal units (Btu). The motivation of this scenario is that legislation prior to the 1990 CAAA [Mitchell S. 1894 (1987); Waxman-Sikorski H.R. 2666 (1987); and Proxmire–Simpson S. 316 (1987)] had default provisions of a uniform standard of emissions by heat input (Regens, 1989). The emission from each plant is equal to the heat input of that plant times the standard. As with the first counterfactual, the aggregate amount of emissions does not change; all that changes is the location of emissions. The intent of these scenarios is to isolate the impact of trading, holding the total amount of $SO_2$ fixed.

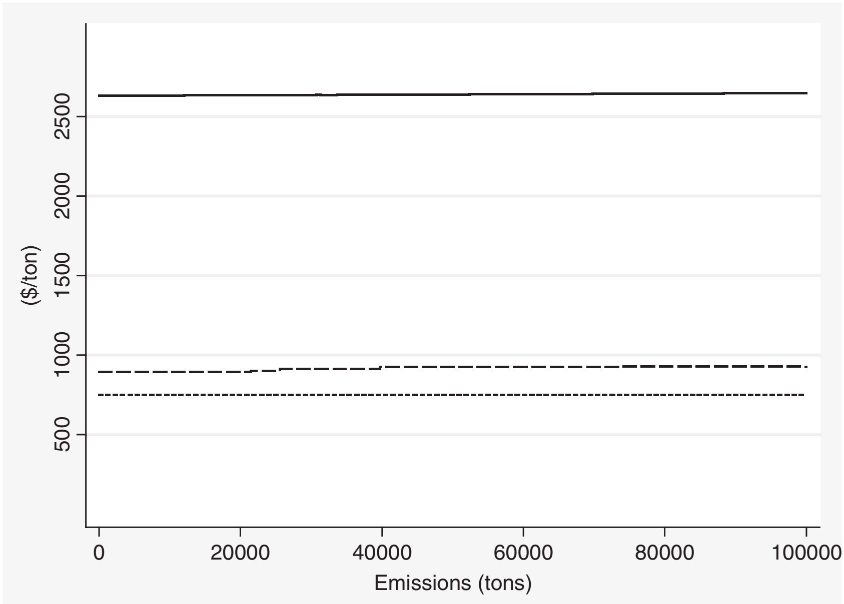
Total damages in each scenario for a specific source ($i$) in a given year ($t$) are computed by taking the product of the emissions ($E_{it}$) times the marginal damage ($MD_i$) of $SO_2$ for that facility. The damages are summed across all 754 generating stations to determine the damage estimate for that year as shown in Equation (1), where $t$ = year, and $i$ = facility.

$$TD_t = (MD_i \times E_{it}) \tag{1}$$

This measures the total damages ($TD_t$) given the scenario emissions ($E_{it}$).

Equation (1) assumes that marginal damages are constant for each facility regardless of the level of emissions. As shown in Figure 6, this assumption is supported by simulated results revealing that the marginal damage of $SO_2$ emitted from each point source is nearly constant over a very broad range of emissions levels (Muller & Mendelsohn, 2009). The primary reason that the marginal damage function is flat is that each source has only a minute impact on ambient concentrations. Shifts in emissions at one facility consequently do not change marginal damages. Of course, that does not mean that marginal damages are the same for each source. Marginal damages are quite different for different sources because a ton from different sources leads to a wide range of human exposures depending on the proximity of large populations.

There are three complications associated with computing the change in damages due to trading relative to the two counterfactuals. First, the aggregate emissions change over time, from 11.2 million tons (2000) to 8.9 million tons (2007). The emissions are not the same in every year because ARP allows facilities to bank allowances for use in a future year. To control for fluctuations in emissions due to




*Key:* Solid line: Damage per ton $SO_2$ emitted in large urban area.
Dash: Damage per ton $SO_2$ emitted in small city.
Dot: Damage per ton $SO_2$ emitted from large electric power generating station.
*Source:* Muller and Mendelsohn (2009).

**Figure 6.** Marginal damage functions for $SO_2$.

banking, we match the total number of emissions in the no-trading cases to the total number of actual emissions in the trading case by proportionally changing the allowance allocation for that year. This effectively removes allowances that are banked for future years. Second, some allowances have been allocated to facilities that emit very little $SO_2$ because they burn natural gas or diesel, or are not operating. We do not count the allowances or emissions from these facilities in either the trade or no-trade case. Third, some facilities contain both coal or residual oil-fired units and natural gas- or diesel-fired units. We examine emissions and heat input only from units that burn coal or residual oil (that emit $SO_2$).

## RESULTS

Table 1 reports the trend in both aggregate emissions and damages over the period from 2000 to 2007. As can be seen in Table 1, the aggregate tons of emissions have been generally falling over time in anticipation of the regulations becoming stricter (column 1). Aggregate damages have therefore been falling over time as well. For example, using the APEEP default assumptions, damages are falling in column 2. Column (3) reports the damages in the first no-trade counterfactual. The damages in the trade case (column 2) are higher than the damages in the no-trade case (column 3). Trading $SO_2$ permits has increased total damages. Table 1 shows that trading has led to an increase in damages of \$0.8 to \$1.1 billion, annually (column 4). This represents a 5 percent to 7 percent increase in damage (column 8).

Further evidence of the adverse impact of trading is evident in columns 5 and 6. Trading has increased the damage per ton of $SO_2$ emitted. The damages with trading are about \$90 per ton more harmful (column 7). Trading is causing this increase

**Table 1.** Effect of trades on damages from $SO_2$ emissions with initial allocation counterfactual.

| | Annual Emissions (million tons $SO_2$) | Total Damages ($billions) | | | Damages per Ton | | | |
|---|---|---|---|---|---|---|---|---|
| Year | (1) Emissions | (2) Observed Emissions | (3) Allowance Allocation | (4) Change | (5) Observed Emissions | (6) Allowance Allocation | (7) Change | (8) Percent Change |
| 2000 | 11.2 | $18.6 | $17.7 | $0.87 | $1,667 | $1,588 | $78 | 4.9% |
| 2001 | 10.6 | $17.6 | $16.7 | $0.83 | $1,664 | $1,585 | $79 | 5.0% |
| 2002 | 10.2 | $17.0 | $16.1 | $0.91 | $1,675 | $1,586 | $90 | 5.7% |
| 2003 | 10.5 | $17.7 | $16.6 | $1.08 | $1,680 | $1,577 | $103 | 6.5% |
| 2004 | 10.2 | $17.1 | $16.0 | $1.08 | $1,680 | $1,574 | $106 | 6.7% |
| 2005 | 10.2 | $17.1 | $16.0 | $1.09 | $1,680 | $1,573 | $107 | 6.8% |
| 2006 | 9.4 | $15.6 | $14.7 | $0.90 | $1,670 | $1,573 | $96 | 6.1% |
| 2007 | 8.9 | $14.9 | $14.1 | $0.86 | $1,677 | $1,580 | $97 | 6.1% |

*Note:* Damages based on default modeling assumptions in APEEP. The no-trade case assumes each plant's emission is equal to the plant's initial allowance allocation.

**Table 2.** Effect of trades on damages from $SO_2$ emissions with the performance standard counterfactual.

| | Annual Emissions (million tons $SO_2$) | Total Damages ($billions) | | | Damages per Ton | | | |
|---|---|---|---|---|---|---|---|---|
| Year | (1) Emissions | (2) Observed Emissions | (3) Uniform | (4) Change | (5) Observed | (6) Uniform | (7) Change | (8) Percent Change |
| 2000 | 11.2 | $18.6 | $17.1 | $1.54 | $1,666 | $1,528 | $138 | 9.0% |
| 2001 | 10.6 | $17.6 | $16.1 | $1.52 | $1,664 | $1,520 | $144 | 9.5% |
| 2002 | 10.2 | $17.0 | $15.4 | $1.63 | $1,675 | $1,515 | $160 | 10.6% |
| 2003 | 10.5 | $17.7 | $15.8 | $1.90 | $1,680 | $1,500 | $181 | 12.1% |
| 2004 | 10.2 | $17.1 | $15.3 | $1.84 | $1,680 | $1,500 | $180 | 12.0% |
| 2005 | 10.2 | $17.1 | $15.3 | $1.78 | $1,680 | $1,505 | $176 | 11.7% |
| 2006 | 9.4 | $15.6 | $14.0 | $1.59 | $1,670 | $1,500 | $170 | 11.3% |
| 2007 | 8.9 | $14.9 | $13.4 | $1.56 | $1,677 | $1,501 | $176 | 11.7% |

*Note:* Damages based on default modeling assumptions in APEEP. The no-trade case employs the uniform performance standard counterfactual.

in damages by inadvertently encouraging $SO_2$ emissions to migrate from low marginal damage (rural) to high marginal damage (urban) sources.

Note that the average damage ($/ton) has remained quite stable from 2000 to 2007. Without trading, damages average $1,580 per ton. With trading, damages average $1,670 per ton. This constancy implies that the observed decrease in total damages over this period has been driven by the tightening cap and the reduction in observed emissions.

Table 2 displays the results of the uniform performance standard counterfactual. The increase in damage from trading is almost twice as great as the increase reported in Table 1. In this case, firms with low emission rates per Btu would have extra allowances to sell, and firms with high emissions rates per Btu would want to buy allowances. Firms might have low emission rates per Btu if they are new and more energy efficient or if they burn low-sulfur fuels (such as Western coal). The fact that the performance standard leads to such a large increase in damages suggests that low emission per Btu plants are likely to be in rural areas and high emission per Btu plants are in urban areas. High emission per Btu plants also had high $SO_2$ marginal damages. The fact that the ARP allowances led to a smaller increase in damages than the performance standard suggests that these high emission per Btu plants managed to get higher initial allowance allocations.

From 2000 to 2007 the increase in damages ranges between $1.5 billion and $1.9 billion, annually (see Table 2). The percentage increase in damages due to trading relative to the performance standard ranges between 9 and 12 percent. Hence, the increase in damages from trading in Table 2 is significantly larger than in Table 1.

In Table 3 we report a sensitivity analysis that relies on the valuation assumptions used by USEPA in their cost–benefit analyses of the Clean Air Act (USEPA, 1999). Instead of a VSL of $2 million, we use a VSL of $6 million. Instead of relying on the remaining years of life, we assume that the VSL is constant for every age group. Both of these new assumptions increase the value placed on mortality risks and increase marginal damages across the board. The additional damages from trading increase by a factor of approximately three in Table 3 compared to Tables 1 and 2 (columns 6 and 7). Trading from 2000 to 2007 increases damage between $2.4 and $3.2 billion annually under the allowance allocation. With the performance standard, trading increases damages by between $4.4 and $5.4 billion annually.

**Table 3.** Effect of trades on damages from $SO_2$ emissions using USEPA valuation of a statistical life.

| Annual Emissions (million tons $SO_2$) | | Total Damages ($billions) | | | Change in Damages ($billions) | |
|---|---|---|---|---|---|---|
| Year | (1) Emissions | (2) Observed Emissions | (3) Allowance Allocation | (5) Uniform Standard | (6) Allowance Allocation | (7) Uniform Standard |
| 2000 | 11.2 | $51.3 | $48.7 | $46.8 | $2.58 | $4.45 |
| 2001 | 10.6 | $48.4 | $45.9 | $44.0 | $2.44 | $4.35 |
| 2002 | 10.2 | $46.8 | $44.2 | $42.2 | $2.59 | $4.59 |
| 2003 | 10.5 | $48.7 | $45.6 | $43.3 | $3.10 | $5.39 |
| 2004 | 10.2 | $47.1 | $44.0 | $41.9 | $3.11 | $5.22 |
| 2005 | 10.2 | $47.0 | $43.9 | $41.9 | $3.15 | $5.10 |
| 2006 | 9.4 | $43.0 | $40.4 | $38.5 | $2.59 | $4.50 |
| 2007 | 8.9 | $41.1 | $38.6 | $36.7 | $2.46 | $4.40 |

## CONCLUSION

As predicted by the theoretical models in the literature (Mendelsohn, 1986), a positive correlation between marginal abatement costs and marginal damages across firms implies that ton-for-ton trading would increase aggregate damages. We argue that the NAAQS cause there to be a positive correlation for $SO_2$. Cities have high densities of polluting sources and so have high pollution concentrations. That leads to nonattainment with the NAAQS, more restrictive emission regulations, and high marginal abatement costs for sources in cities. Higher wages and rents also contribute to higher marginal abatement costs in cities. Urban areas also have high concentrations of people, which cause emissions to lead to high exposures, large health effects, and therefore high marginal damages. Cities tend to be nonattainment areas, and they tend to have high marginal damages. Damages and abatement costs are positively correlated, and ton-for-ton trading inadvertently causes emissions to move from rural areas to urban areas, increasing overall damages.

We measure the magnitude of this increased damage using the results of an integrated assessment model. The model is used to estimate the marginal damages of emissions from every power plant regulated under the ARP. We then examine how trading affects where emissions are located. We start by assuming that emissions would have occurred where allowances were initially given if there had been no trades. We also examine an alternative initial distribution of emissions that would have resulted from using a performance standard based on emission rates per Btu. We contrast these initial allocations against the actual emissions that occurred with trading. In all cases, the marginal damage at each plant does not change. In all cases, the aggregate amount of emissions is the same. The only difference across cases is the location of emissions. The location changes with each trade.

The magnitude of the increased damages varies slightly over the time period examined. It is likely that as the cap on $SO_2$ continues to decrease, the magnitude of the increased damages from trading will fall, as there will be fewer trades. Of course, the abatement cost savings will also fall. The magnitude of the damages depends on the counterfactual: how emissions would have been distributed if trading was not allowed. We look at both the initial distribution of allowances and a performance standard as two prominent alternatives. The magnitude of the damages also depends on which values are used to weigh health effects. Because the trading leads to a substantial increase in health effects, the higher the value used, the greater are the damages.

This paper demonstrates that the cap and trade policy governing $SO_2$ emissions under the ARP has caused the total damage due to $SO_2$ emissions to increase relative to both the initial allowance allocation and to an allocation based on performance standards. We examine all of the 754 original power plants regulated under the 1990 CAAA (99.8 percent of total emissions). Between 2000 and 2007, a conservative value of mortality risks suggests that trading from the initial allowances increased damages between \$0.8 and \$1.1 billion annually. With the USEPA preferred valuation of mortality risk, the increase in damages is between \$2.4 and \$3.2 billion annually. Assuming allowances were initially allocated on the basis of a performance standard, a conservative value of mortality risks suggests that trading increased damages in this period between \$1.5 and \$1.9 billion annually. With the USEPA values, this same trading increased damages between \$4.4 and \$5.4 billion annually. In contrast, the literature estimates that the abatement cost savings from trading over this period are between \$0.7 and \$2.1 billion annually (Carlson et al., 2000; Keohane, 2006; Ellerman et al., 2000). The increased damages are at least as large as the cost savings, and they may well be larger. Trading $SO_2$ on a ton-for-ton basis appears not to have increased welfare compared to command and control regulations.

If trading were instituted for other criteria pollutants, such as particulates, very similar results would likely occur for the same reasons. First, damages per unit emission of local pollutants tend to be higher in cities because of the importance of proximal population densities. Second, urban counties tend to be in nonattainment; consequently, firms in these counties face higher marginal abatement costs. However, the negative impact of ton-for-ton trading on damages would not necessarily apply to all pollutants. For example, greenhouse gas pollutants have the same marginal damage in all locations, so there is no spatial correlation between marginal damages and marginal abatement costs. There would be no change in global warming damages associated with the cap and trade of greenhouse gases.

The policy implication of this paper is not to ban cap and trade entirely. The results argue that trading regimes for criteria pollutants need to be modified. Instead of trading on a ton-for-ton basis, pollutants should be traded on a marginal damage basis. The government should weight tons of emissions by marginal damage and then allow trading of the weighted tons. This would equilibrate marginal damage to marginal cost at each location and yield an efficient result (Montgomery, 1972; Baumol & Oates, 1988; Farrow et al., 2005; Muller & Mendelsohn, 2009).

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## REFERENCES

Baumol, W., & Oates, W. (1988). The theory of environmental policy, 2nd ed. Cambridge: Cambridge University Press.

Burtraw, D., & Mansur, E. (1999). Environmental effects of $SO_2$ trading and banking. Environmental Science and Technology, 33, 3489–3494.

Burtraw, D., Krupnick, A., Mansur, E., Austin, D., & Farrell, D. (1998). Costs and benefits of reducing air pollutants related to acid rain. Contemporary Economic Policy, 26, 379–400.

Byun, D., & Schere, K. (2006). Review of the governing equations, computational algorithms, and other components of the Models-3 Community Multiscale Air Quality (CMAQ) modeling systems. Applied Mechanics Reviews, 59, 51–77.

Carlson, C., Burtraw, D., Cropper, M., & Palmer, K. (2000). Sulfur dioxide control by electric utilities: What are the gains from trade? Journal of Political Economy, 108, 1292–1326.

Chestnut, L., & Rowe, R. (1990). Preservation values for visibility protection at national parks. Report No. EE-0405. Research Triangle Park, NC: USEPA Office of Air and Radiation.

Dales, J. (1968). Pollution, property, and prices. Toronto: University of Toronto Press.

Ellerman, A., Joskow, P., Schmalensee, R., Montero, J., & Bailey, E. (2000). Markets for clean air. New York: Cambridge University Press.

Farrow, R., Schultz, M., Celikkol, P., & Van Houtven, G. (2005). Pollution trading in water quality limited areas: Use of benefits assessment and cost effective trading ratios. Land Economics, 81, 191–205.

Hausker, K. (1992). The politics and economics of auction design in the market for sulfur dioxide pollution. Journal of Policy Analysis and Management, 11, 553–572.

Keohane, N. (2006). Cost savings from allowance trading in the 1990 Clean Air Act: Estimates from a choice-based model. In C. E. Kolstad & J. Freeman (Eds.), Moving to markets in environmental regulation: Lessons from twenty years of experience (pp. 194–229). New York: Oxford University Press.

Mendelsohn, R. (1986). Regulating heterogeneous emissions. Journal of Environmental Economics and Management, 13, 301–312.

Montgomery, W. (1972). Markets in licenses and efficient pollution control programs. Journal of Economic Theory, 5, 395–418.

Mrozek, J., & Taylor, L. (2002). What determines the value of life? A meta-analysis. Journal of Policy Analysis and Management, 21, 253–270.

Muller, N., & Mendelsohn, R. (2007). Measuring the damages from air pollution in the United States. Journal of Environmental Economics and Management, 54, 1–14.

Muller, N., & Mendelsohn, R. (2009). Efficient pollution regulation: Getting the prices right. American Economic Review, 99, 1714–1739.

Muller, N., Mendelsohn, R., & Nordhaus, W. (in press). Environmental accounting for pollution: Methods with an application to the U.S. economy. American Economic Review.

Muller, N., Tong, D., & Mendelsohn, R. (2009). Regulating $NO_x$ and $SO_2$ in Atlanta. B. E. Journal of Economic Analysis & Policy, 9, Article 3.

Nordhaus, W. (2000). From porcopolis to carbopolis: The evolution from pork bellies to emissions trading. In R. F. Kosobud (Ed.), Emissions trading (pp. 61–73). New York: John Wiley.

Pope, C., Burnett, R., Thun, M., Calle, E., Krewski, D., & Ito, K., et al. (2002). Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. Journal of the American Medical Association, 287, 1132–1141.

Regens, J. (1989). Acid rain policymaking and environmental federalism: Recent developments, future prospects. Publius, 19, 75–91.

Schmalensee, R., Joskow, P., Ellerman, A., Montero, J., & Bailey, E. (1998). An interim evaluation of sulfur dioxide emissions trading. Journal of Economic Perspectives, 12, 53–68.

Shadbegian, R., Gray, W., & Morgan, C. (2007). Benefits and costs from sulfur dioxide trading: A distributional analysis. In G. E. Visgilio & D. M. Whitelaw (Eds.), Acid in the environment (pp. 241–259). New York: Springer.

Tietenberg, T. (1980). Transferable discharge permits and the control of stationary source air pollution: A survey and synthesis. Land Economics, 56, 391–416.

Turner, D. (1994). Workbook of atmospheric dispersion estimates: An introduction to dispersion modeling, 2nd ed. Ann Arbor, MI: Lewis Publishers.

U.S. Environmental Protection Agency. (1999). The benefits and costs of the Clean Air Act: 1990–2010. EPA Publication No. 410-R-99-001. Washington, DC: U.S. Government Printing Office.

U.S. Environmental Protection Agency. (2006). National emissions inventory. Washington, DC: Office of Air Quality Planning and Standards, Emissions Inventory Group, Emissions, Monitoring, and Analysis Division.

U.S. Environmental Protection Agency. (2010a). Clean air markets—Data and maps. Retrieved March 5, 2010, from http://camddataandmaps.epa.gov/gdm/.

U.S. Environmental Protection Agency. (2010b). The green book nonattainment areas for criteria pollutants. Retrieved February 24, 2010, from http://www.epa.gov/oaqps001/greenbk/mapnpoll.html.

Viscusi, W., & Aldy, J. (2003). The value of a statistical life: A critical review of market estimates throughout the world. Journal of Risk Uncertainty, 27, 5–76.