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A Spatial Econometric Approach to Measuring Pollution Externalities: An Application to Ozone Smog

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Abstract. This paper uses spatial econometrics to analyze air pollution externalities. State-by-state source-receptor transfer coefficients that can be used as a basis for a location-differentiated permit system are estimated. Results affirm the importance of regional transport in determining local ozone air quality, although owing to non-monotonicities in ozone production the externality is not always negative. Because the origin of emissions matters, results also reject a non-spatially differentiated NO_x cap and trade program as an appropriate mechanism for reducing ozone smog.

1. Introduction

In 1997, eight states in the northeastern United States filed petitions under Section 126 of the Clear Air Act, claiming that emissions from upwind states were affecting their ability to attain and maintain the Environmental Protection Agency (EPA) National Ambient Air Quality Standard (NAAQS) for ozone smog. These petitions identified 31 states plus the District of Columbia as containing sources that significantly contribute to the regional transport of ozone (EPA, 1999b; Helms, 2002).[1] All the petitions target sources in the Midwest; some of the petitions also target sources in the south, southeast, and northeast (EPA, 1999a). Were these petitions justified? Is it indeed the case that emissions from one state may affect the air quality in another state?

The principal ingredient of smog, tropospheric ozone (O_3) is the most difficult to control of the six criteria pollutants for which National Ambient Air Quality Standards have been established (Chang and Suzio, 1995). Among ozone's adverse effects on humans are labored breathing, impaired lung functions, increased hospital admissions and emergency room visits for respiratory causes, and possible long-term lung damage. Ozone exposures have also been associated with a wide range of vegetation effects, including visible foliar injury, growth reductions and yield loss in agricultural crops, growth reductions in seedlings and mature trees, and impacts at forest stand and ecosystem levels (EPA, 1997b; Sillman, 1995a).

The original primary and secondary National Ambient Air Quality Standards for ozone established in 1979 were each a 1-hour average of 120 parts per billion (ppb), not to be exceeded more than three times in three years. In July 1997, based on its review of the available scientific evidence linking ozone exposures to adverse health and welfare effects at levels allowed by the 1-hour standards, the EPA revised both ozone standards to 8-hour standards at levels of 80 ppb, with forms based on the three-year average of the annual fourth-highest daily maximum 8-hour average ozone concentrations measured at each monitor within an area (EPA, 1998). In March 2008, EPA again strengthened the 8-hour ozone standard to 75 ppb (EPA, 2009).

A secondary pollutant, ozone is not emitted directly but is formed in ambient air by chemical reactions involving nitrogen oxides (NO_x), which consist of nitrogen oxide (NO) and nitrogen dioxide (NO_2), and volatile organic compounds (VOCs). NO_x is emitted from fossil fuel combustion, biomass burning, lightning, stratospheric flux, and microbial activity in soils (Carroll and Thompson, 1995), while VOCs are

[1] According to the EPA (1999b), only 30 states plus DC were identified.

emitted from combustion, industry and vegetation (NRC, 1992). Cities with high emission rates, warm temperatures, frequent inversions, and stagnant meteorology are most vulnerable to high levels of ozone smog (Sillman, 1993).

The rate of ozone production shows a nonlinear and non-monotonic dependence on precursor concentrations. There are two different photochemical regimes: a NO_x-limited regime, in which the rate of ozone formation increases with increasing NO_x and is insensitive to changes in VOC; and a VOC-limited regime, in which the rate of ozone formation increases with increasing VOC and may even decrease with increasing NO_x (Sillman, 1999). Thus, higher emissions of NO_x do not always result in higher levels of ozone pollution; in some cases, higher NO_x emissions may actually *decrease* ozone, a phenomenon known as NO_x titration.[2]

Both ozone and its precursors are transboundary pollutants. As a consequence, individual cities do not always have direct control of their own attainment of the ozone standard. For instance, according to the EPA (1997a, II.B.4): "a reduction in transport into the New York area associated with upwind emissions reductions on the order of 75 percent for NO_x and 25 percent for VOC along with local VOC and NO_x reductions may be needed for attainment in New York."

To assess the extent of regional transport, the EPA has relied primarily on the simulation results of atmospheric chemistry models.[3] For example, the 1990 Clean Air Act Amendments require the use of 3-D Eulerian photochemical modeling for planning ozone attainments in many nonattainment areas (Chang and Suzio, 1995; Sistla et al., 1996). Previous studies analyzing the effects of transport have relied on atmospheric chemistry models as well. Cleveland and Graedel (1979) use chemical kinetic modeling to analyze the effect of emissions and transport on air quality in the New Jersey–New York City metropolitan region. Cleveland et al. (1976) use an air parcel trajectory model to analyze the effects of emissions in the New York City metropolitan area on air quality in Connecticut and Massachusetts.

While these models incorporate natural phenomena such as wind patterns, seasonal cycles, chemical processes, and biological emissions, they have several drawbacks. First, the models do not generate standard errors for their estimates. These estimates are the result of many functional form and parameter assumptions that are made in order to specify the equations governing chemical processes and transport. For example, rate constants are assumed to be a given function of temperature and other factors, and natural emissions of isoprene are assumed to be a parametric function of a given set of base emissions. While many of these functional form and parameter assumptions may have been derived from actual data or experiments, and therefore should have confidence intervals associated with them, they are instead treated as if they were known with certainty.

A second drawback with using models to measure transport is that the models are deterministic. In contrast, since its formation requires sunlight, ozone smog is in part a function of stochastic factors such as weather. It is unclear whether these model simulations appropriately handle the stochastic component to ozone formation.

A third problem with the photochemical models is that their accuracy is limited. For example, uncertainties in boundary conditions (Winner et al., 1995) and in meteorological parameters such as wind fields and mixing heights (Sistla et al., 1996) cast doubt on the accuracy of VOC-NO_x sensitivity predictions (Chang and Suzio, 1995; Sillman, 1995b). Models can also err in their prediction of sensitivity because similar ozone concentrations can be produced in either VOC- or NO_x-sensitive environments (Sillman, 1995b).

A fourth problem with the atmospheric chemistry models is that supporting data for input and diagnostic evaluations are sparse or lacking for most regions (Blanchard et al., 1999). A fifth problem is that models are computationally expensive and costly in terms of both time and money (Blanchard et al., 1999; Winner et al., 1995).

The purpose of this paper is to measure regional transport using a different approach from atmospheric chemistry modeling: spatial econometrics. This paper teases out, statistically, the extent to which precursor emissions from one location impose an externality on ozone air quality in another state. Spatial econometrics is used in several ways. First, this paper tests for spatial autocorrelation in ozone and in its precursors. After confirming the spatial autocorrelation, it is then determined if the spatial autocorrelation is due the transport of emissions from elsewhere, or if the spatial autocorrelation is instead due merely to spatial autocorrelation of omitted variables such as climate, industrial patterns or exogenous shocks. This paper then examines the geographical extent of transport to determine if air quality at one site is affected by emissions from hundreds of kilometers away. This paper

[2] For a scientific explanation of NO_x titration, see Lin (2000).

[3] The majority of models are Eulerian models, which simulate the concentration and transport of air pollution at every grid point and time step. Another type of model is a Lagrangian model, which follows a given air parcel, but must make the assumption that each air parcel is independent and therefore that there are no interactions between air parcels.

next tests for whether a non-spatially differentiated NO_x cap and trade program can reduce ozone smog. Lastly, to form the basis for a spatially differentiated cap and trade program instead, this paper estimates state-by-state source-receptor transfer coefficients that measure, for each state, the effect of emissions from that state on air quality in the other states.

A spatial econometric approach to measuring air pollution externalities has several advantages over the conventional modeling approach. First, by estimating reduced-form relationships between emissions and air quality at neighboring sites, one can avoid having to make any of the parametric, structural or functional form assumptions that are needed for an atmospheric chemistry model – assumptions that can sometimes be ad hoc. Second, the use of econometrics yields confidence intervals for the estimates, and therefore provides a more informative measure of the externality and its significance. Third, an alternative means of measuring air pollution externalities enables us to compare the validity of the modeling and econometric approaches. Fourth, one can use econometrics to test whether a NO_x cap and trade program is an appropriate mechanism for reducing local levels of ozone smog.

This paper is important for several reasons. First, the use of spatial econometrics rather than atmospheric chemistry models to analyze emissions transport is a methodological contribution. Second, methods that account for the spatial dimension of social, economic and environmental processes are of statistical and econometric interest. Third, externalities are an important concept in economics and especially in environmental economics; this paper quantifies air pollution externalities. Fourth, the results have important implications for policy, especially those involving regional coordination. An example of such a policy is a regional NO_x cap and trade program. This paper tests to see if a non-spatially differentiated NO_x cap and trade program amongst multiple states would be an appropriate mechanism for reducing ozone pollution. If not, the source-receptor transfer coefficients estimated in this paper can form the basis for a spatially-differentiated permit system instead.

The results affirm the importance of transport in determining local air quality, although owing to non-monotonicities in ozone production the externality is not always negative. Because the origin of emissions matters, results also reject a non-spatially differentiated NO_x cap and trade program as an appropriate mechanism for reducing ozone smog.

The balance of this paper proceeds as follows. Section 2 describes the data and tests for spatial autocorrelation in ozone and in its precursors. After confirming the spatial autocorrelation, Section 3 analyzes whether the spatial autocorrelation in the data arises from true or spurious state dependence. In other words, it determines whether the spatial autocorrelation is due the transport of emissions from elsewhere, or if the spatial autocorrelation is instead due merely to spatial autocorrelation of omitted variables such as climate, industrial patterns or exogenous shocks. Section 4 examines the geographical extent of transport to determine if air quality at one site is affected from emissions hundreds of kilometers away. Section 5 tests for whether a non-spatially differentiated NO_x cap and trade program can reduce ozone smog. To form the basis for a spatially differentiated cap and trade program instead, Section 6 estimates state-by-state source-receptor transfer coefficients that measure, for each state, the effect of emissions from that state on air quality in the other states. Section 7 concludes.

2. Data

Hourly ozone concentration data for July 1990 were extracted from the EPA's Aerometric Information Retrieval System (AIRS). At each AIRS monitoring site, ozone is measured by Advanced Pollution Instrumentation (API) 400 ozone analyzers which are subject to frequent calibration and consistency checks. Automated zero (0 ppb), precision (100 ppb) and level 1 (400 ppb) checks are performed at least every three days, level checks are performed twice a month, and full calibration occurs every three months (Lin, 2000). Most ozone instruments at AIRS sites measure within 5-6% of the true value most of the time for ozone concentrations in the range of 30-80 ppb, with the accuracy somewhat higher for ozone concentrations in higher ranges (Fairley, 1999).

In accordance with the protocol for the new NAAQS, running 8-hour averages, indexed by the first hour, were calculated for each 8-hour interval with at least 6 hours of data, and the daily maximum 8-hour average was stored for each day for each site.

Daily maximum temperature data were extracted from National Climatic Data Center (NCDC) Summary of the Day First Order data files. These files contain daily selected elements of observations taken by certified stations operated by the National Weather Service, United States Air Force, United States Navy, and the Federal Aviation Administration (NCDC, 1998). Following Fiore et al. (1998), the NCDC sites were selected for the length of their records and their proximity to AIRS sites. One NCDC site was selected for each 4° latitude by 5° longitude (approximately 400 km (250 miles) by 500 km (310 mi.)) grid square. A

finer resolution is unnecessary because the lifetime of ozone is long enough for ozone and temperature to be correlated throughout the same grid square, and because small-scale variations in temperature and ozone are not necessarily correlated (Lin et al., 2001). These assumptions are verified in Lin (2000).

Annual 90th percentile 1-hour ozone data in parts per billion (ppb) were obtained via monitor data queries of the EPA's Air Quality System (AQS) database.[4] Annual county-level NO_x and VOC emissions estimates data for 1990, 1996-1999 and 2001 were obtained from the EPA's National Emission Inventory.[5] These estimates account for emissions from both mobile and point sources, including fuel combustion from electric utilities, industrial and other sources, solvents, on-road vehicles, and non-road engines and vehicles. Data collected by various federal, state and local agencies are used to calculate the total emissions for each pollutant. For example, to estimate emissions from fossil fuel fired steam electric utilities, the EPA uses information on fuel consumption, emissions factors, fuel characteristics, and control efficiency based on monthly boiler-level data collected by the Department of Energy's Energy Information Administration. The EPA's estimation methodology is explained in detail in EPA (2001).

The annual county-level population and per capita personal income (nominal $) variables are taken from the Bureau of Economic Analysis (BEA)'s County Summary CA1-3 1969-2001 (BEA, 2003a).[6] Per capita income data were deflated to 1982-1984 U.S. dollars using the consumer price index (CPI). County population data were converted to population per square mile using county area data from the EPA.

Table 1 presents summary statistics for the data on ambient ozone concentrations, NO_x and VOC emissions, population, income, and temperature. Figure 1 presents maps of the variables, with observations color-coded by quartile. Dotted lines at 36°N in latitude and 97.5°W in longitude divide the continental U.S. into four quadrants: Northeast (NE), Northwest (NW), Southeast (SE) and Southwest (SW).

There are several key features of these maps. First, it seems that many of the variables have a spatial component; for ozone, for example, high values tend to be clustered in the Midwest, along the East coast, and in California. Second, the spatial patterns for precursor emissions are not identical to the ones for ozone air quality. For example, in the eastern half of the US, high values of NO_x emissions extend further west than high values of 90th percentile ozone do, suggesting that emissions of NO_x might be transported eastward.

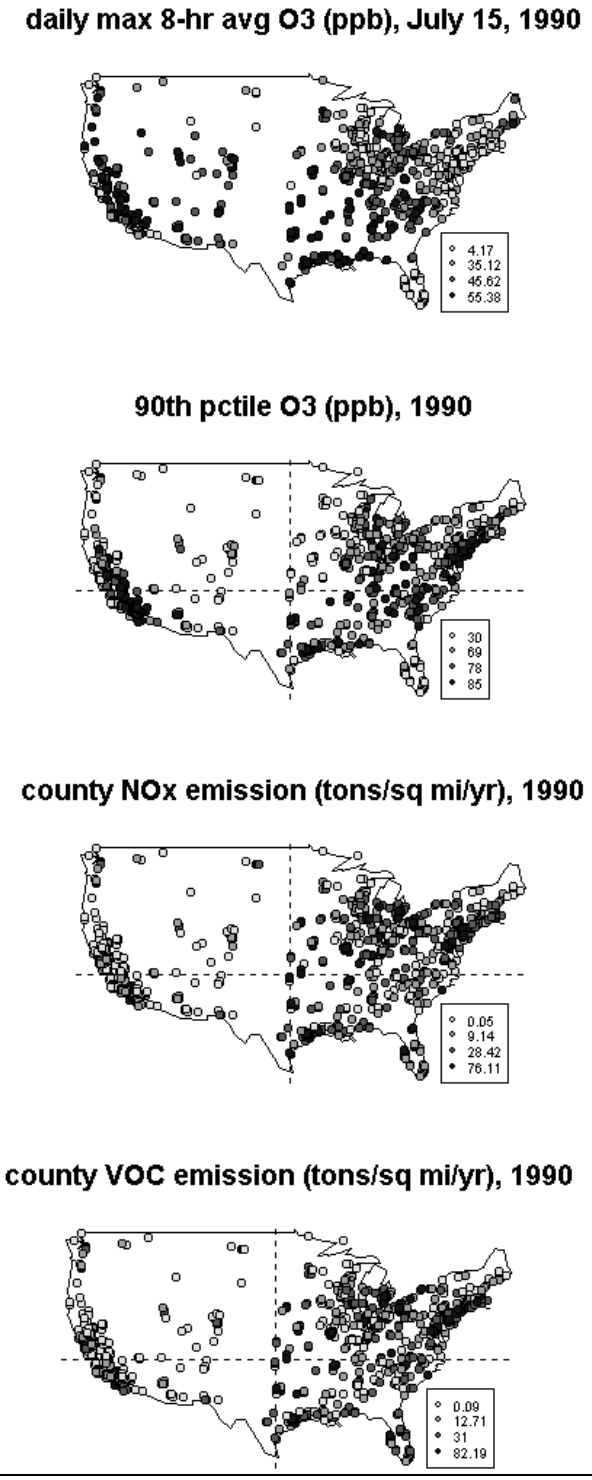

Figure 1. Quartile maps.

[4] www.epa.gov/aqspubl1/annual_summary.html.

[5] These were the years for which emissions data were available. Owing to funding constraints faced by the EPA, the data for 1991-1995 are currently inconsistent and therefore could not be used. I thank Tom McMullen from the EPA for extracting the data for me.

[6] For more details about the BEA data, see BEA (2003b).

Table 1. Summary statistics.

Variable	**# obs**	**mean**	**s.d.**	**min**	**max**
daily maximum 8-hour average ozone (ppb)	23422	52.53	21.57	0.12	190.0
annual 90th percentile ozone (ppb)	6310	77.33	13.49	27.00	190.0
county NO_x emissions (tons per square mile per year)	6310	56.33	133.49	0.015	2657.4
county VOC emissions (tons per square mile per year)	6310	52.14	136.99	0.010	3287.9
county population per sq mi	6310	994.79	3565.4	0.006	67348
county per capita real income (1000 1982-1984 $)	6310	15.02	4.72	0.015	52.504
daily maximum temperature (K)	23422	303.76	5.09	285.78	318.56

To test for spatial autocorrelation in the ozone, precursor emissions and control variables, the Moran's *I* test and the Geary's *c* test are used. For both tests, the null hypothesis H_0 of no spatial autocorrelation is tested against the two-sided alternative hypothesis H_1 that the data are spatially autocorrelated. Two versions of the null hypothesis H_0 are used: one under a normality assumption and the other under a randomization assumption. Under the normality assumption for H_0, the observed map is assumed to be the result of *n* independent draws from a normal population, and is therefore one possible realization of an underlying normal probability model. Under the randomization assumption for H_0, the observed map is one possible arrangement of the set of *n* values (Haining, 1990). For both tests, a spatial neighborhood is defined using distance (Bivand and Portnov, 2002). In particular, the first-order "neighbors" of any particular site *i* consist of all other sites with data in the given year located between 1 km and 500 km (in Great Circle distance) from site *i*.

According to the results of the tests for spatial autocorrelation, for nearly all variables and for both types of tests, and for both distributional assumptions, the null hypothesis of no spatial autocorrelation can be rejected at a 5% significance level.[7] Thus, ozone and its precursors exhibit spatial autocorrelation.

3. Spurious or true state dependence?

This section examines whether the spatial autocorrelation in ambient ozone is due to transport or merely to omitted variables. Is air quality at one location affected by the transport of emissions from elsewhere, or is the spatial autocorrelation due merely to spatial autocorrelation of omitted variables such as climate, industrial patterns or exogenous shocks?

In the unrestricted version of the model, which is termed the *spatial distance model*, air quality *z*, a vector in which each component z_i is the air quality of site *i*, is given by:

$$z = \rho Wz + X\beta_1 + WX\beta_2 + \varepsilon, \quad (1)$$

where *W* is a weight matrix, *X* is a matrix of explanatory variables including emissions, and ε is i.i.d. normal. The weight matrix *W* is derived from assuming, as was done above, that the first-order "neighbors" of any particular site *i* consist of all other sites located between 1 km and 500 km from site *i*, and from requiring the weights to all of site *i*'s neighbors sum to 1 for each site *i*. *Wz* is termed the "distanced" value of *z*, as it represents the value of *z* at neighboring sites. The parameter ρ indicates the extent of spatial interaction between neighboring observations.

As seen from its reduced-form version,

$$z = (I - \rho W)^{-1}(X\beta_1 + WX\beta_2 + \varepsilon), \quad (2)$$

the spatial distance model implies that the air quality z_i at any one site depends on the distanced values of the explanatory variables X_i. In particular, if *X* includes emissions, then this means that air quality at one site depends on the emissions from a neighboring site. If the autocorrelation in air quality *z* does indeed arise from a dependence on distanced emissions *X*, then, following the terminology used by Heckman (1978) in an analogous time series context, there is true state dependence.

However, if the following restriction holds:

$$\beta_2 = -\rho\beta_1 \quad (3)$$

then the model collapses to:

$$z = X\beta_1 + \mu, \quad (4)$$

[7] All the p-values are < 2.2E-16, with the exception of VOC emissions for Geary's *c* under randomization (p-value = 0.119); NO_x emissions for Geary's *c* under randomization (p-value =0.0051); and population for Geary's *c* under both normality (p-value = 4.076E-10) and randomization (p-value = 0.3054).

where μ exhibits spatial autocorrelation, since

$$\mu = (I - \rho W)^{-1}\varepsilon, \quad (5)$$

where ε is i.i.d. normal. In this restricted version of the model in equation (4), which is termed the *spatial error model*, the spatial dependence comes from autocorrelation in the error term, and not from dependence of air quality z on distanced emissions X. The spatial autocorrelation in the error term may arise through omitted variables that have a spatial dimension, such as climate, industrial patterns, or exogenous shocks (Abreu, de Groot and Florax, 2004).[8] If the underlying model is the spatial error model, then, to borrow Heckman's (1978) terminology, there is spurious, not true, state dependence.

Thus, if the autocorrelation is in the error term, then there is spurious state dependence; if there is dependence of air quality z on distanced emissions X, then there is true state dependence. It is now tested whether the spatial dependence comes from autocorrelation in the error term, or from dependence of z on distanced X.

In order to test for true versus spurious state dependence, the parameters in each of the two models – the unrestricted spatial distance model and the restricted spatial error model – are estimated via a two-step procedure. In the first step, for each value of ρ over a grid of possible values, the coefficients $\beta_1^c(\rho)$, $\beta_2^c(\rho)$, the sum of squared residuals $SSR(\rho)$, and the concentrated log likelihood

$$\mathrm{L}^c(\rho) = -n\ln\left(\frac{SSR(\rho)}{n}\right) + \ln(\det(\mathrm{I} - \rho W)) - \frac{n}{2} \quad (6)$$

are estimated via OLS. In the second step, the concentrated log likelihood $L^c(\rho)$ is maximized over ρ.

For the unrestricted spatial distance model, the concentrated log likelihood is obtained by running OLS of $(z - \rho Wz)$ on $[X\ WX]$. For the restricted spatial error model, the concentrated log likelihood is obtained by running OLS of $(z - \rho Wz)$ on $(X - \rho WX)$.

The two-step procedure is conducted twice: first optimizing over ρ from a coarse grid over the range $[-4\rho_o, 6\rho_o]$, where ρ_o is the estimate of ρ obtained from an initial estimation of the spatial distance model in equation (1). The optimum ρ_{coarse} of the coarse grid is then used to form a fine grid $[-\rho_{coarse}, 2\rho_{coarse}]$ over which to obtain the estimated optimum $\hat{\rho}_{opt} \equiv \rho_{fine}$. Both grids consist of 100 points each. Estimates of the remaining coefficients are then obtained by re-running OLS with $\hat{\rho}_{opt}$ to obtain $\hat{\beta}_1 \equiv \beta_1^c(\hat{\rho}_{opt})$ and $\hat{\beta}_2 \equiv \beta_2^c(\hat{\rho}_{opt})$.[9]

The 95% confidence interval $[\rho_{low}, \rho_{high}]$ for $\hat{\rho}_{opt}$ is obtained from using the inverse log likelihood ratio test to find ρ_{low} and ρ_{high} that satisfy:

$$2\left(L^c(\hat{\rho}_{opt}) - L^c(\rho_{low})\right) = \chi_{crit} \quad (7)$$

and

$$2\left(L^c(\hat{\rho}_{opt}) - L^c(\rho_{high})\right) = \chi_{crit} \quad (8)$$

respectively, where χ_{crit} is the critical value of the chi-squared distribution at 95% and with 1 degree of freedom.

True versus spurious state dependence is tested for using a likelihood ratio test. If the null hypothesis that the restrictions hold is rejected at a 5% significance level, this means that the spatial autocorrelation is from dependence of air quality z on distanced emissions X rather than from spatial autocorrelation in the error term, i.e., that there is true, not spurious, state dependence. Otherwise, the dependence is merely spurious.

Table 2 presents the results for the regressions using the daily maximum 8-hour average ozone as the dependent variable z. The unit of observation is an ozone monitoring site on a given day. The table reports the coefficients from the unrestricted spatial distance model as well as the result of the test for spurious versus true state dependence. The following are used for the dependent variables X: the previous day's maximum 8-hour average ozone (ppb), county NO_x emissions (tons per square mile per year), county VOC emissions (tons per square mile per year), county population per square mile, county per capita personal income (1982-1984 $), and daily maximum temperature (K).

High levels of ozone correlate strongly with high temperatures in the observations (NRC, 1991), reflecting the effects of biogenic VOC emissions, temperature-driven chemistry, and air-mass stagnation (Jacob

[8] Owing to spatial autocorrelation in the error term, estimation of the spatial error model via OLS results in unbiased but inefficient estimates (Abreu et al., 2004).

[9] Standard errors under this OLS need to be corrected to account for the derivative of $\det(I - \rho W)$.

et al., 1993; Sillman and Samson, 1995). Put another way, the total derivative dO_3/dT describes a sum of partial derivatives $(\partial O_3/\partial x_i)(\partial x_i/\partial T)$, where $\mathbf{x} = (x_1, x_2, \ldots x_n)'$ is the ensemble of ozone forcing variables that are temperature-related. Lin et al. (2001) previously found a strong relationship between daily maximum temperature and the probability of exceeding the 8-hour average air quality standard of 80 ppb in different regions of the United States. Many previous studies have employed observed relationships between ozone and meteorological variables to make short-term forecasts (or hindcasts) of ozone air quality (e.g., Cox and Chu, 1993; Eshel and Bernstein, 2006; Ordonez et al., 2005). By controlling for temperature in the spurious vs. true state dependence analysis, most of the weather patterns that might provide a plausible alternative explanation for the presence of spatial correlations in ozone levels are therefore controlled for.

There are two key features of the results to note. First, the estimated $\hat{\rho}$, which measures the extent of spatial interaction between neighboring observations, is significant and positive for all quadrants. Second, all quadrants except the Southwest exhibit true state dependence. Thus, transport of emissions is an important determinant of ozone in all regions except the heavily-polluted Los Angeles basin, which is characterized by stagnant meteorological conditions, slow winds and temperature inversion that limit the dispersion rate of the pollutants, as well as by a clean upwind environment that minimizes long-distance transport into the city (Sillman, 1993).

Table 2. Spurious vs. true state dependence.

Dependent variable is daily maximum 8-hour average ozone (ppb)				
	NE	NW	SE	SW
previous day's maximum 8-hour ozone (ppb)	0.31 ***	0.71 ***	0.51 ***	0.68 ***
	(0.02)	(0.02)	(0.03)	(0.02)
county NO_x emissions (tons per square mile per year)	0.00	-0.01	-0.02	0.25 .
	(0.00)	(0.03)	(0.02)	(0.13)
county VOC emissions (tons per square mile per year)	0.00	0.03	0.05	-0.05
	(0.01)	(0.03)	(0.03)	(0.05)
county population (100 per square mile)	-0.05 *	-0.17 *	-0.22	-0.86
	(0.02)	(0.07)	(0.22)	(0.58)
county per capita income (1000 1982-1984 $)	-0.17 **	-0.34 **	-0.10	-0.16
	(0.05)	(0.13)	(0.24)	(0.26)
daily maximum temperature (K)	1.55 ***	0.28 *	1.47 ***	0.08
	(0.09)	(0.12)	(0.36)	(0.14)
p-value (Pr>F)	0.00 ***	0.00 ***	0.00 ***	0.00 ***
adj. R^2	0.27	0.63	0.29	0.69
# observations	2410	730	782	759
95% confidence interval for ρ	[1.02, 1.26]	[0.72, 0.92]	[0.84, 0.99]	[0.48, 0.74]
spurious or true state dependence	TRUE	TRUE	TRUE	SPURIOUS

Notes: Standard errors in parentheses. The unit of observation is an ozone monitoring site on a given day. Daily data from July 5, 10, 15, 20, 25, and 30, 1990 are used. A constant is also included in the models. Coefficients and standard errors are from the initial estimation of the unrestricted spatial distance model; the p-value and adjusted R^2 are from the final OLS estimation. A likelihood ratio test at a 5% significance level is used to determine spurious or true state dependence. Significance codes: * 5% level, ** 1% level, *** 0.1% level.

4. Regional pollution transport

Having determined that, except in the Southwest, spatial autocorrelation in air quality is due to pollution transport and not to omitted variables, this section examines the geographical extent of transport. Is air quality at one site affected by emissions from hundreds of kilometers away?

A spatial non-simultaneous autoregressive lag model with multiple spatial orders is run on the daily panel data. The different spatial orders are neighbors of different distances. The first-order spatial distances of any particular site *i* consist of all other sites located between 1 km and 500 km from site *i*, and the second-order spatial distances consist of sites located between 500 km and 1000 km from site *i*. These two orders

correspond roughly to the intrastate scale and the interstate scale, respectively.

In particular, the daily maximum 8-hour average ozone at a particular day and location is regressed on the previous day's maximum 8-hour average ozone at the same location, from a first-order distance away and from a second-order distance away, and on emissions from first- and second-order distances away. The unit of observation is an ozone monitoring site on a given day. The following controls are used: population density, income, and temperature, as well as dummies for quadrant, state, county, and day.

Table 3. Geographical extent of transport (dependent variable is daily maximum 8-hour average ozone (ppb)).

previous day's maximum 8-hour average ozone (ppb)		
same location	0.44	***
	(0.01)	
1 km to 500 km away	-0.05	***
	(0.01)	
500 km to 1000 km away	-0.04	**
	(0.01)	
county annual NOx emissions (tons per square mile)		
same location	0.00	
	(0.00)	
1 km to 500 km away	-0.51	***
	(0.10)	
500 km to 1000 km away	-0.46	***
	(0.11)	
county annual VOC emissions (tons per square mile)		
same location	-0.00	
	(0.01)	
1 km to 500 km away	0.36	***
	(0.09)	
500 km to 1000 km away	0.29	**
	(0.10)	
controls		
population per square mile	-0.00	
	(0.00)	
income (1000 1982-1984 $)	0.36	***
	(0.17)	
daily maximum temperature (K)	1.71	***
	(0.03)	
p-value (Pr>F)	0.00	***
adj. R^2	0.56	
# obs	23421	
p-value from test that distanced emissions are not needed	[0.00]	***
p-value from joint test of all 2nd distances	[0.00]	***

Notes: Standard errors in parentheses. The unit of observation is an ozone monitoring site on a given day. Daily data for July 1990 are used. Controls also include dummies for quadrant, state, county, and day. Significance codes: * 5% level, ** 1% level, *** 0.1% level.

According to the results in Table 3, distanced emissions do matter, even after accounting for time lagged air quality. Thus, emissions from up to 1000 km away can affect local air quality. First- and second-order distanced NO_x emissions have a negative effect on ozone concentrations, thus improving air quality, likely because of NO_x titration. In contrast, both first- and second-order distanced VOC emissions have a positive effect on ozone concentrations, thus worsening air quality. For both NO_x and VOCs, the magnitudes of

the coefficients on the first-order distanced emissions are greater than those on the second-order distanced emissions, so that intrastate transport has a greater impact on local air quality than does interstate transport. As expected, higher temperatures correspond with higher ozone levels; this is consistent with previous studies (see, e.g., Lin et al., 2001).

5. Cap and trade

A non-spatially differentiated NO_x cap and trade program amongst multiple states would be an appropriate mechanism for reducing ozone pollution if it did not matter to ambient ozone concentrations whence each ton of NO_x was emitted. A ton of NO_x emitted from Indiana should have the same effect on air quality as a ton of NO_x emitted from Kentucky; only the total quantity of NO_x emitted should matter. To determine whether such a cap and trade program is appropriate for the a group of states, the model where the NO_x emissions are disaggregated by state is tested against one in which the NO_x emissions are aggregated over all the states in the group. This test is conducted for two groups of states that have considered cap and trade programs: the states in the Ozone Transport Commission and the states in the 1998 NO_x SIP call.

The Ozone Transport Commission (OTC) is comprised of the following states: Maine, New Hampshire, Vermont, Massachusetts, Connecticut, Rhode Island, New York, New Jersey, Pennsylvania, Maryland, Delaware, the northern counties of Virginia, and the District of Columbia. In 1994, the OTC adopted a memorandum of understanding (MOU) to achieve regional emission reductions of NO_x. States signing the MOU were committed to developing and adopting regulations that would reduce region-wide NO_x emissions in 1999 and further reduce emissions in 2003 (EPA, 2004b).[10]

In September 1998, in effort to mitigate the regional transport of ground-level ozone in the eastern half of the United States, the EPA finalized a rule, known as the NO_x SIP call, that required 22 states and the District of Columbia to submit state implementation plans (SIPs) to reduce NO_x emissions (EPA, 2002). These states are: Alabama, Connecticut, Delaware, District of Columbia, Georgia, Illinois, Indiana, Kentucky, Maryland, Massachusetts, Michigan, Missouri, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Rhode Island, South Carolina, Tennessee, Virginia, and West Virginia.[11] Under the NO_x SIP call, the EPA developed the NO_x Budget Trading Program to allow states to meet their emission budgets in a cost-effective manner through participation in a region-wide non-spatially differentiated cap and trade program. As of the 2007 ozone season, all affected states and the District of Columbia chose to meet most of their NO_x SIP call requirements through participation in the NO_x Budget Trading Program (EPA, 2009).

The model with the aggregated emissions, i.e., without distanced-emissions effects, was rejected at a 5% significance level. Thus, a non-spatially differentiated NO_x cap and trade program is not appropriate for reducing ozone for either the states in the OTC or the states in the NO_x SIP call. Policymakers should use a spatially-differentiated program, for example one that takes into account the state-by-state source-receptor coefficients estimated in the next section of this paper, instead. The results are consistent with the atmospheric chemistry modeling results of Mauzerall et al. (2005). In contrast to a cap and trade program, which presumes that shifts of emissions over time and space, holding the total fixed over the course of the summer ozone season, will have minimal effect on the environmental outcome, Mauzerall et al. (2005) show that a shift of a unit of NO_x emissions from one place or time to another could result in large changes in resulting health effects due to ozone formation and exposure.

6. Source-receptor coefficients

The results of the previous section suggest that a non-spatially differentiated cap and trade program is inappropriate for reducing ozone smog, and that a spatially differentiated one should be used instead. Such a program would take into account the different impacts that emissions from each state have on air quality in each other state.

In particular, state-by-state source-receptor transport coefficients that measure how much an increase in NO_x emissions from a given source state affects ozone air quality in a receptor state are estimated. State-by-state source-receptor transport coefficients are important for two reasons. First, the transport of air pollution becomes even more important if the pollutant crosses the border between different air quality management jurisdictions, since then the pollution control policy in one jurisdiction imposes externalities on another jurisdiction. These source-receptor

[10] Virginia was not a signatory of the MOU. The OTC NO_x Budget Program ran from 1999 to 2002 and is now replaced by the NO_x SIP call (EPA, 2004b).

[11] Wisconsin was removed via court order. Georgia is not listed on: http://www.dep.state.wv.us/item.cfm?ssid=8&ss1id=295 but Georgia's website does mention NOx SIP call: www.air.dnr.state.ga.us/sspp/noxsipcall/

coefficients measure how emissions in one state affect air quality in other states. Second, these source-receptor coefficients can form the basis for a spatially-differentiated cap and trade program.

To estimate the source-receptor coefficients, a separate regression is run for each state of the annual 90th percentile ozone at a particular air quality monitoring site in that state on the total annual NO_x emissions from own and neighboring states. Because prevailing winds between 30°N and 50°N in latitude blow from the Southwest (Jacob, 1999), and because emissions from up to 1000 km (620 mi.) away can affect local air quality, state *j* is considered a neighbor of state *i* if it is located within 1000 km either to the South, West or Southwest of state *i*. The unit of observation is an ozone monitoring site in a given year. County population density and county per capita income are controlled for. The controls reduce the possibility of spatial autocorrelation in the error term due to omitted variables that have a spatial dimension.

The state-by-state source-receptor transport coefficients measure how an additional 1000 tons of NO_x emissions in one state affects the 90th percentile ozone level in a downwind state. The individual coefficients are available in the appendix[12]. For instance, if Ohio emitted an additional 100,000 tons of NO_x over the course of one year, which is less than 10% of its average annual emission of 1.17 million tons, the annual 90th percentile ozone level in Michigan would increase by a statistically significant 17 ppb. Some of the statistically significant source-receptor coefficients are negative; this is likely due to the non-monotonic nature of ozone formation.

One main advantage of the spatial econometric approach over the atmospheric chemistry modeling approach is that the estimates from the former approach have standard errors associated with them, and it is therefore possible to assess whether certain effects are statistically significant. For instance, while they both have positive source-receptor transfer coefficients, neither the impact of NO_x emissions from Illinois on air quality in Indiana, nor the impact of NO_x emissions from Ohio on air quality in New Jersey is statistically significant; without the standard errors, one may have mistakenly interpreted the effects to be positive.

Table 4 reports, for each source state, the total net effect of NO_x emissions from that state, as measured by the sum of its effect on air quality in all of its receptor states, including itself. Only coefficients that are significant at a 5% level are included in calculating the total net effect. Each of the total net values is an estimate of the impact of an additional 1000 tons of emissions in a particular state on ozone exposure

Table 4. Total net effect of source-receptor coefficients.

State	Mean Annual Nox Emissions (1000 tons)	Total Effect on 90th percentile Ozone
Alabama	581.52	1.23
Arizona	403.95	1.07
Arkansas	303.69	-3.03
California	1413.24	-0.11
Colorado	339.65	-1.94
Connecticut	154.71	-1.41
Delaware	70.12	2.58
DC	16.26	17.04
Florida	990.31	1.07
Georgia	704.26	2.67
Idaho	128.77	-0.25
Illinois	1012.02	-0.1
Indiana	838.76	-2.05
Iowa	327.8	4.17
Kansas	418.12	-0.04
Kentucky	685.91	0.13
Louisiana	835.99	-2
Maine	91.33	
Maryland	356.78	0.46
Massachusetts	324.25	
Michigan	849.79	-0.02
Minnesota	471.08	
Mississippi	378.35	0.05
Missouri	578.99	
Montana	185.21	0.55
Nebraska	231.69	0
Nevada	141.65	0.8
New Hampshire	82.01	
New Jersey	437.41	-0.44
New Mexico	326.5	2.08
New York	856.36	0.14
North Carolina	672.63	
North Dakota	206.87	
Ohio	1165.16	0.33
Oklahoma	438.42	2.24
Oregon	243.29	0.29
Pennsylvania	981.25	0.22
Rhode Island	37.29	
South Carolina	361.99	
South Dakota	96.78	
Tennessee	661.92	
Texas	1943.47	0.07
Utah	256.18	-0.04
Vermont	43.18	
Virginia	555.22	
Washington	343.76	0.04
West Virginia	481.74	
Wisconsin	461.95	
Wyoming	268.95	

Notes: For each source state, the total net effect of NO_x emissions from that state on 90th percentile ozone is the sum of its effect on air quality in all of its receptor states, including itself. Only coefficients significant at a 5% level are included in calculating the total effect.

[12] The coefficients with standard errors can also be found online at www.des.ucdavis.edu/faculty/Lin/airqual_ext_AppA.pdf.

throughout the rest of the country. These estimates could be used in the design of efficient environmental regulation, which would equate the marginal damage of pollution to marginal abatement costs across space (Muller and Mendelsohn, forthcoming). For example, the resulting ratios of these estimates could be used as a starting point for the determination of a location-differentiated permit system. These estimates could therefore have a significant impact on policy.

7. Conclusion

This paper uses spatial econometrics to analyze air pollution externalities. Results affirm the importance of regional pollution transport in determining local ozone air quality. However, the transport of NO_x can sometimes be a positive externality rather than a negative one; this is likely due to non-monotonicities in ozone production.

General features of the spatial econometric results are consistent with atmospheric science and with the results of atmospheric chemistry models. Ozone exhibits spatial correlation and, except in the Los Angeles basin, as is consistent with the science, this correlation is due to pollution transport rather than simply to spatially correlated omitted variables. NO_x and VOC emissions from up to 1000 km away have significant effects on ambient ozone concentrations. High temperature is correlated with high ozone levels.

The spatial econometric approach improves upon the atmospheric chemistry modeling approach because its estimates have standard errors associated with them, because it does not make prior assumptions on the parameters, and because spatial econometric models are less computationally expensive and take less time to run. Moreover, the spatial econometric approach yields a test for the appropriateness of a non-spatially differentiated NO_x cap and trade program as well as state-by-state source-receptor transfer coefficients that can be used as a basis for a location-differentiated permit system.

Cap and trade programs have been used to decrease pollution in a variety of contexts. In the 1980s, a cap and trade program was used to facilitate the phase-out of stratospheric ozone-depleting chloroflourocarbons. In the 1990s, a cap-and-trade program was adopted to reduce sulfur dioxide emissions and consequent acid rain by 50 percent under the Clean Air Act amendments of 1990. Most recently, cap and trade programs have emerged as the preferred national and regional policy instrument to address carbon dioxide emissions linked with global climate change (Stavins, 2009). These non-spatially differentiated cap and trade system are appropriate for decreasing the target pollutant – whether it be choroflourocarbons, sulfur dioxide or carbon dioxide – because the source of the emissions did not matter. Only the overall quantity of the pollutant mattered to overall damages.

Similarly, a non-spatially differentiated NO_x cap and trade program amongst multiple states would be an appropriate mechanism for reducing ozone pollution if it did not matter to ambient ozone concentrations whence each ton of NO_x was emitted. A ton of NO_x emitted from Indiana should have the same effect on Connecticut's air quality as a ton of NO_x emitted from Kentucky; only the total quantity of NO_x emitted should matter. However, results show that is not the case: the location of NO_x emissions does matter to overall ozone air quality. As a consequence, a non-spatially differentiated cap and trade program is not appropriate for either the states in the OTC or the states in the NO_x SIP call as a mechanism for reducing ozone smog. Unlike cap and trade programs for choroflourocarbons, sulfur dioxide or carbon dioxide, a program that aims to decrease ozone pollution by capping and trading NO_x pollution permits would need to be spatially differentiated in order to be effective.

Results of this paper, particularly the state-by-state source-receptor transfer coefficients, have important implications for policy. Because NO_x emissions in one state can affect the ozone air quality in other states, a regional approach to ozone smog control is needed. Moreover, rather than use a non-spatially differentiated NO_x cap and trade program to reduce ozone smog, policymakers should use a spatially differentiated program, for example one that takes into account the state-by-state source-receptor coefficients estimated in this paper, instead.

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Appendix: Source-Receptor Coefficients

This table presents the results of state-by-state regressions of the annual 90th percentile ozone (ppb) in the receptor state on annual NO_x emissions (1000 tons) from own and neighboring states. The unit of observation is an ozone monitoring site in a given year. Controls also include population and income. Only coefficients that are significant at a 5% level are included in calculating the total net effect. Significance codes: * 5% level; ** 1% level; *** 0.1% level.

NO_x	O_3	O_3 in:								
from:	total:	AL	AZ	AR	CA	CO	CT	DE	DC	FL
AL	1.23	0.70							0.24 ***	
AZ	1.07		0.02 *							
AR	-3.03	-0.94		0.62 ***		-3.02 ***				
CA	-0.11		0.00		0.00	-0.10 ***				
CO	-1.94					-2.80 ***				
CT	-1.41						0.55 **			
DE	2.58						-0.83 ***	3.58 *		
DC	17.04						7.50 ***	-8.47	4.87 ***	
FL	1.07									0.21 ***
GA	2.67							0.82 **	0.37 ***	
ID	-0.25									
IL	-0.10								0.00	
IN	-2.05						-0.24 ***	-0.99 **	-0.32 ***	
IA	4.17									
KS	-0.04									
KY	0.13						0.02 *	0.40 **		
LA	-2.00	-0.75 **		-0.67 ***						0.16 ***
ME										
MD	0.46									
MA										
MI	-0.02									
MN										
MS	0.05	1.08								-0.30 ***
MO										
MT	0.55									
NE	0.00									
NV	0.80		0.01			0.66 ***				
NH										
NJ	-0.44									
NM	2.08			-0.05		6.82 ***				
NY	0.14									
NC										
ND										
OH	0.33									
OK	2.24	-0.06		0.88 ***						
OR	0.29									
PA	0.22									
RI										
SC										
SD										
TN										
TX	0.07			0.01						0.00
UT	-0.04									
VT										
VA										
WA	0.04									
WV										
WI										
WY										

NO_x	O_3	O_3 in:								
from:	total:	GA	ID	IL	IN	IA	KS	KY	LA	ME
AL	1.23	0.19			-1.13			0.12 *		
AZ	1.07		0.00		0.17					
AR	-3.03	-0.26 *		-0.11 ***		-0.14 ***	1.25 ***	-0.23 ***		
CA	-0.11		-0.02 ***							
CO	-1.94					0.08 **	1.49 ***			
CT	-1.41									1.03
DE	2.58									-1.98
DC	17.04									0.38
FL	1.07	0.20			-0.65			0.17 ***		
GA	2.67	-0.03						-0.04		
ID	-0.25		-0.24 ***							
IL	-0.10			0.02 *	0.19			0.00		
IN	-2.05									
IA	4.17			0.85 ***	3.48	1.01 ***				
KS	-0.04			-0.06 ***		-0.09 ***	-0.19 ***			
KY	0.13			-0.03 **						
LA	-2.00	-0.06				-0.03			0.02	
ME										-0.66
MD	0.46									
MA										-1.22
MI	-0.02									
MN										
MS	0.05									
MO										
MT	0.55									
NE	0.00									
NV	0.80		0.02							
NH										
NJ	-0.44									
NM	2.08						-3.28 ***		-0.08 ***	
NY	0.14									
NC										
ND										
OH	0.33									
OK	2.24						0.41 ***			
OR	0.29		0.29 ***							
PA	0.22									
RI										
SC										
SD										
TN										
TX	0.07								0.01	
UT	-0.04									
VT										
VA										
WA	0.04									
WV										
WI										
WY										

Significance codes: * 5% level; ** 1% level; *** 0.1% level.

NO_x from:	O_3 total:	O_3 in: MD	MA	MI	MN	MS	MO	MT	NE	NV
AL	1.23	1.26 ***		-0.38 ***						
AZ	1.07							0.12 ***	0.22 ***	
AR	-3.03				-0.13 ***	0.56 *	-0.26 ***			
CA	-0.11							0.01 ***		0.00
CO	-1.94				0.10 **		-0.02	-0.12 ***	0.01	
CT	-1.41		-1.44 ***							
DE	2.58		0.53 *							
DC	17.04									
FL	1.07									
GA	2.67	1.48 ***								
ID	-0.25							0.01	0.08 ***	
IL	-0.10	-0.07 ***		-0.03 ***						
IN	-2.05	-0.63 ***	0.28 ***	-0.08 ***						
IA	4.17				0.96 ***					
KS	-0.04				-0.10 ***		0.08 **		-0.05	
KY	0.13	-0.21 ***	-0.05 ***							
LA	-2.00					-0.66 **	-0.08 **			
ME										
MD	0.46		0.24 ***							
MA										
MI	-0.02			-0.02 *						
MN					-0.05					
MS	0.05					0.03	0.35 ***			
MO										
MT	0.55							0.13 ***		
NE	0.00								-0.36 ***	
NV	0.80									0.00
NH										
NJ	-0.44									
NM	2.08									
NY	0.14									
NC										
ND										
OH	0.33			0.17 ***						
OK	2.24					0.87 **				
OR	0.29									
PA	0.22									
RI										
SC										
SD										
TN										
TX	0.07					0.01				
UT	-0.04									
VT										
VA										
WA	0.04									
WV										
WI										
WY										

Significance codes: * 5% level; ** 1% level; *** 0.1% level.

NO_x	O_3	O_3 in:								
from:	total:	NH	NJ	NM	NY	NC	ND	OH	OK	OR
AL	1.23					0.10 **		0.14		
AZ	1.07			0.13 **					0.19 ***	
AR	-3.03					-0.24 ***		-0.24 ***		
CA	-0.11			-0.01						0.01
CO	-1.94						-0.46 ***			
CT	-1.41	-0.63								
DE	2.58	0.49 ***	-2.34 ***		0.66 ***					
DC	17.04		4.67 ***							
FL	1.07					0.17 ***		0.18 ***		
GA	2.67				-0.05	-0.05		-0.04		
ID	-0.25						-0.12 ***			
IL	-0.10									
IN	-2.05									
IA	4.17				0.50 ***					
KS	-0.04						0.23 ***			
KY	0.13									
LA	-2.00									
ME										
MD	0.46	0.20								
MA		0.07								
MI	-0.02	-0.04								
MN										
MS	0.05									
MO										
MT	0.55						0.28 ***			
NE	0.00						0.36 ***			
NV	0.80			0.14 ***						0.11
NH										
NJ	-0.44		-0.18 **		-0.26 ***					
NM	2.08			-0.42 ***					-0.16 ***	
NY	0.14				0.14 ***					
NC						0.01				
ND										
OH	0.33		0.01					0.00		
OK	2.24								0.08 *	
OR	0.29									-0.05
PA	0.22		0.23 ***							
RI										
SC										
SD										
TN										
TX	0.07			0.03 *					-0.02 *	
UT	-0.04									
VT										
VA										
WA	0.04									
WV										
WI										
WY										

Significance codes: * 5% level; ** 1% level; *** 0.1% level.

NO_x	O_3	O_3 in:								
from:	total:	PA	RI	SC	SD	TN	TX	UT	VT	VA
AL	1.23	-0.35 ***		0.19		0.19				0.12 *
AZ	1.07				0.11 ***					
AR	-3.03			-0.26 *		-0.26 *	0.26 ***	0.06		-0.23 ***
CA	-0.11							0.00		
CO	-1.94				-0.23 ***					
CT	-1.41		-0.52 ***						-0.63	
DE	2.58		0.39						0.49 ***	
DC	17.04		0.90							
FL	1.07			0.20		0.20				0.17 ***
GA	2.67			-0.03		-0.03				-0.04
ID	-0.25				-0.02 *					
IL	-0.10	-0.04 ***								0.00
IN	-2.05	-0.07 ***								
IA	4.17									
KS	-0.04				0.09 ***					
KY	0.13									
LA	-2.00			-0.06		-0.06				
ME										
MD	0.46		0.22 *						0.20	
MA									0.07	
MI	-0.02		-0.04						-0.04	
MN										
MS	0.05									
MO										
MT	0.55				0.14 ***					
NE	0.00									
NV	0.80							0.02		
NH										
NJ	-0.44									
NM	2.08						-0.39 ***			
NY	0.14									
NC										
ND										
OH	0.33	0.16 ***								
OK	2.24									
OR	0.29									
PA	0.22	-0.01 *								
RI										
SC										
SD										
TN										
TX	0.07						0.06 ***			
UT	-0.04							-0.04 *		
VT										
VA										
WA	0.04									
WV										
WI										
WY										

Significance codes: * 5% level; ** 1% level; *** 0.1% level.

NO_x from:	O_3 total:	O_3 in: WA	WV	WI	WY
AL	1.23		0.12 *		
AZ	1.07				0.28 ***
AR	-3.03		-0.23 ***	-0.11 ***	
CA	-0.11	0.00			0.00
CO	-1.94				0.08
CT	-1.41				
DE	2.58				
DC	17.04				
FL	1.07		0.17 ***		
GA	2.67		-0.04		
ID	-0.25				0.05 ***
IL	-0.10		0.00	0.02 *	
IN	-2.05				
IA	4.17			0.85 ***	
KS	-0.04				
KY	0.13			-0.03 **	
LA	-2.00				
ME					
MD	0.46				
MA					
MI	-0.02				
MN					
MS	0.05				
MO					
MT	0.55				
NE	0.00				
NV	0.80	0.08			
NH					
NJ	-0.44				
NM	2.08				-0.41 ***
NY	0.14				
NC					
ND					
OH	0.33				
OK	2.24				
OR	0.29	-0.04			
PA	0.22				
RI					
SC					
SD					
TN					
TX	0.07				
UT	-0.04				
VT					
VA					
WA	0.04	0.04 **			
WV					
WI					
WY					

Significance codes: * 5% level; ** 1% level; *** 0.1% level.