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Using Spatial Econometrics to Measure Ozone Pollution Externalities

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ABSTRACT

This paper summarizes my previous work in Lin (2010), in which I use spatial econometrics to analyze air pollution externalities. In Lin (2010), 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.

Keywords: Air Pollution; Externalities; Spatial Econometrics

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 [1,2].[1] All the petitions target sources in the Midwest; some of the petitions also target sources in the south, southeast, and northeast [3]. 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 is the most difficult to control of the six criteria pollutants for which National Ambient Air Quality Standards have been established [4]. 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 [5,6].

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 [7], while VOCs are emitted from combustion, industry and vegetation [8]. Cities with high emission rates, warm temperatures, frequent inversions, and stagnant meteorology are most vulnerable to high levels of ozone smog [9].

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 [10]. 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: "a reduction in transport into the New York area associated with upwind emissions reductions on the order of 75 percent

[1]According to [1], only 30 states plus DC were identified.

[2]For a scientific explanation of NO_x titration, see [11].

for NO_x and 25 percent for VOC along with local VOC and NO_x reductions may be needed for attainment in New York" [12].

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 [4,13]. Previous studies analyzing the effects of transport have relied on atmospheric chemistry models as well. Reference [14] uses chemical kinetic modeling to analyze the effect of emissions and transport on air quality in the New Jersey-New York City metropolitan region. Reference [15] uses 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 [16] and in meteorological parameters such as wind fields and mixing heights [13] cast doubt on the accuracy of VOC-NO_x sensitivity predictions [4,17]. Models can also err in their prediction of sensitivity because similar ozone concentrations can be produced in either VOC- or NO_x-sensitive environments [17].

A fourth problem with the atmospheric chemistry models is that supporting data for input and diagnostic evaluations are sparse or lacking for most regions [18]. A fifth problem is that models are computationally expensive and costly in terms of both time and money [16,18].

This paper summarizes my previous work in [19], in which I measure regional transport using a different approach from atmospheric chemistry modeling: spatial econometrics. Reference [19] 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, [19] 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. Reference [19] then examines the geographical extent of transport to determine if air quality at one site is affected by emissions from hundreds of kilometers away. Reference [19] 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, [19] 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.

My work in [19] 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,

[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.

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. Reference [19] 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.

2. Spurious or True State Dependence?

Reference [19] first 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?

There are two key features of the results to note. First, the parameter 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 [9].

3. Regional Transport

Having determined that, except in the Southwest, spatial autocorrelation in air quality is due to transport and not to omitted variables [19], then 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 distances is run on the daily panel data. The different spatial distances 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.

According to the results in [19], 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., [20]).

4. 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. Reference [19] conducts this test 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 com-

mitted to developing and adopting regulations that would reduce region-wide NO_x emissions in 1999 and further reduce emissions in 2003 [21].[4]

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 [22]. 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.[5] 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 [23].

According to the results in [19], the model with the aggregated emissions was rejected at a 5% significance level for both the OTC and the NO_x SIP call. 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 [24]. 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, [24] 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.

5. 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 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, [19] runs a separate regression 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 [25], and because emissions from up to 1000 km 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 an online appendix.[6] 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

[4]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 [21].

[5]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 NO_x SIP call: http://www.air.dnr.state.ga.us/sspp/noxsipcall/

[6]http://www.des.ucdavis.edu/faculty/Lin/airqual_ext_AppA.pdf

New Jersey is statistically significant; without the standard errors, one may have mistakenly interpreted the effects to be positive.

Reference [19] 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 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 [26]. 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.

6. Conclusions

My work in [19] uses spatial econometrics to analyze air pollution externalities. Results affirm the importance of regional 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 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 chlorofluorocarbons. In the 1990's, 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 [27]. These non-spatially differentiated cap and trade system are appropriate for decreasing the target pollutant—whether it be chlorofluorocarbons, 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 chlorofluorocarbons, 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 [19], 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 [19], instead.

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