DEFENSIVE INVESTMENTS AND THE DEMAND FOR AIR QUALITY: EVIDENCE FROM THE NOX BUDGET PROGRAM

By

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# Defensive Investments and the Demand for Air Quality: Evidence from the NOx Budget Program[1]

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ABSTRACT

The demand for air quality depends on health impacts and defensive investments, but little research assesses the empirical importance of defenses. A rich quasi-experiment suggests that the Nitrogen Oxides ($NO_x$) Budget Program (NBP), a cap-and-trade market, decreased $NO_x$ emissions, ambient ozone concentrations, pharmaceutical expenditures, and mortality rates. The annual reductions in pharmaceutical purchases, a key defensive investment, and mortality are valued at about \$800 million and \$1.1 billion, respectively, suggesting that defenses are over one-third of willingness-to-pay for reductions in $NO_x$ emissions. Further, estimates indicate that the NBP's benefits easily exceed its costs and that $NO_x$ reductions have substantial benefits.

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## I. Introduction

Willingness to pay (WTP) for wellbeing frequently depends on factors that enter the utility function directly (e.g., the probability of mortality, school quality, local crime rates, etc.) and compensatory investments that help to determine these factors (Grossman 1972). In a wide variety of contexts, the empirical literature has almost exclusively focused on the direct effects (e.g., health outcomes) of these factors and left the defensive investments largely unmeasured. As examples, there has been little effort to measure: the use of medications or air filters to protect against poor air quality (e.g., Chay and Greenstone 2003; Currie and Neidell 2005); parental expenditures on supplemental tutoring to improve educational outcomes for their children; or the costs of alarm systems and additional security to protect against crime. All of these defensive investments are costly and displace consumption of utility-generating goods. Indeed, economic theory suggests that these actions constitute a significant portion of the costs of harms, as individuals should set the marginal utility of their purchase equal to the marginal utility of avoiding the harm itself. It therefore seems reasonable to presume that the available estimates of willingness to pay for a wide variety of factors are substantially understated.

This paper develops a measure of willingness to pay for air quality improvements that accounts for both the direct health impacts and defensive investments. As a measure of defensive behavior, we investigate whether medication usage responds to changes in air quality. This is likely to be an especially important measure of defensive expenditures, because, for example, the annual cost of prescription medications for asthma is reported to exceed the monetized value of any other component of asthma's social cost, including mortality, emergency department admissions, or lost productivity (Weiss and Sullivan 2001). We also provide new evidence on how air pollution affects more commonly studied outcomes like mortality and hospitalizations.

The empirical application is based on a quasi-experiment that exploits three sources of variation in the introduction of an emissions market for nitrogen oxides ($NO_x$). The $NO_x$ Budget Trading Program (NBP) operated a cap-and-trade system for over 2,500 electricity generating units and industrial boilers in the Eastern and Midwestern U.S. between 2003 and 2008. Because this market had the goal of decreasing ozone pollution, which reaches high levels in summer, the market operated only between May 1 and September 30. Specifically, we use a triple-difference estimator that compares pollution, defensive expenditures and health outcomes in the NBP participating and non-participating states, before versus after 2003, and summer versus winter.[2]

The empirical analysis produces several key results. First, there was a substantial decline in air pollution emissions and ambient concentrations. Figure 1 illustrates the dramatic effect of this market on $NO_x$ emissions in the states participating in the NBP.[3] In 2001-2002, daily $NO_x$ emissions were fairly flat throughout the calendar year, with a peak in summer. In 2005-2007, $NO_x$ emissions were nearly 40% lower but almost entirely during the summer months when the NBP was in force. $NO_x$ emissions are a primary ingredient in the complex function that produces ozone air pollution, so it is unsurprising that we find that the large reductions in $NO_x$ led to declines in mean ozone concentrations of roughly 6% and reduced the number of summer days with high ozone levels (i.e., more than 65 ppb) by about 35%, or a third of a standard deviation.

Second, these improvements in air quality produced substantial benefits. Medication expenditures decreased by about 1.5% or roughly $800 million annually in the 19 Eastern and Midwestern United States where the NBP was in force; this is close to an upper bound estimate of the NBP's total abatement costs. This decline in medication expenditures is evident both among short-acting respiratory medications taken in response to the presentation of respiratory symptoms and long-term control medications that are taken to prevent these episodes. Since people can engage in other defensive investments like avoiding time outdoors or purchasing air filters, medication expenditures provide a lower bound on the total defensive costs associated with air pollution. Further, the summertime mortality rate declined by up to 0.5%, corresponding to 2,200 fewer premature deaths per summer in the NBP states, mainly among individuals 75 and older. The application of age-adjusted estimates of the value of a statistical life (Murphy and Topel 2006) implies this reduced mortality is valued at about $1,100 million annually. The

[2] "Winter" in this paper refers to the combined months of January-April and October-December.
[3] Unless otherwise noted, our data on $NO_x$ emissions refer to emissions from power plants covered by our data (i.e., in the Acid Rain program).

mortality estimates are less precise than the medication ones, and the results must be interpreted accordingly. Additionally, there is little systematic evidence of an effect of the NBP on hospitalization charges. Overall, it is striking that defensive investments account for more than one-third of our estimate of total willingness-to-pay for reductions in $NO_x$ emissions.[4]

Third, the paper provides the first instrumental variables estimates of the effect of $NO_x$ emissions on health and defensive investments. Such information is an essential determinant of air quality policy since $NO_x$ is a pollutant that can be controlled directly by regulation, whereas ozone cannot be. Ambient ozone is determined by a complex function based on several factors including $NO_x$ emissions, volatile organic compound emissions, and temperature. These estimates suggest a significant causal link between $NO_x$ emissions, medication purchases, and mortality. For example, we find that a 10 percentage point (pp) reduction in $NO_x$ emissions leads to 0.06 pp reduction in medication purchases and also a 0.07 pp reduction in mortality. Further, it may be appropriate to conclude the reductions in ozone concentrations stemming from the reductions in $NO_x$ emissions are the primary channel for these health improvements; we cautiously report instrumental variable estimates that reveal positive relationships between ambient ozone concentrations and medication expenditures and mortality rates, respectively.

In addition to providing new evidence on the empirical importance of defensive expenditures in the context of air pollution, this paper makes several contributions.[5] First, the results may be useful for the ongoing and contentious academic and policy debates about the regulation of $NO_x$ emissions as a means to reduce ambient ozone concentrations. The recent

[4] $NO_x$ emissions can influence crop yields (through ozone), visibility, the value of outdoor activities, the purchase of air filters, and other factors. A complete measure of willingness to pay for reductions in $NO_x$ emissions, as well as the defensive expenditures' (e.g., air filter purchases) share, would include all of these factors.

[5] An emerging empirical literature aims to measure behavioral responses, including defenses, to health-reducing environmental factors (Graff-Zivin and Neidell 2009; Neidell 2009; Deschênes and Greenstone 2011; Graff-Zivin, Neidell, and Schlenker 2011; Barreca et al. 2016; Ito and Zhang 2016). An older theoretical literature analyzes defenses and willingness to pay (Courant and Porter 1981; Bartik 1988). A small epidemiological literature, largely using samples of under 100 asthma patients, shows that asthmatics increase medication use on polluted days (Menichini and Mudu 2010). As we discuss later, our focus on summer versus rest-of-year seasonal expenditures improves over existing work, which focuses on same-day effects. Same-day regressions can suffer from short-term displacement if pollution merely shifts the day on which a person uses medication but does not change total medium-run medication use. At the same time, we note that medications may be stored across seasons.

controversy surrounding Volkswagen's use of a "defeat device" that understated $NO_x$ emissions and ongoing litigation about the health consequences reflects the dearth of reliable information on the health consequences of $NO_x$ emissions. More broadly, ozone is one of the six "criteria" pollutants that the Clean Air Act targets, but unlike the other pollutants it has remained persistently high. Further, the Obama Administration tightened the national ambient air quality standard in 2015 from 75 to 70 ppb, following a long political and legal battle between the White House, EPA, Congress, and industry; as of 2015, 126 million Americans or about 40% of the population live in areas that violate this new air quality standard for ozone. These ozone standards are contentious at least partly because much of the previous evidence comes from observational studies where there is a substantial risk of confounding air pollution and other determinants of health.[6] The central role of $NO_x$ emissions in controlling ozone concentrations is underscored by the fact that the regulatory impact analysis for the new ozone standard requires a 65% reduction in $NO_x$ emissions between 2011 and 2025 (USEPA 2015). The paper speaks directly to this debate and can contribute to the regular updating of cost-benefit analyses of the Clean Air Act.[7]

Second, this study is an important step forward in moving closer to the ideal of credibly measuring the consequences of *sustained* exposure to air pollution. Much of the literature relating human health and ozone concentrations focuses on daily or weekly variation in ozone and on specific states or groups of cities; studies based on daily and/or weekly variation are especially subject to concerns about "harvesting" or temporal displacement of mortality (and medication expenditures) and that the true loss of life expectancy is small (Deschenes and

[6] Key papers about the relationship between health and ozone include Bell et al. (2004), Currie and Neidell (2005), NRC (2008), Jerrett et al. (2009), Neidell (2009), Lleras-Muney (2010), Moretti and Neidell (2011), and Dominici et. al (2014).

[7] The results are also important because they fill a knowledge gap in recent research on the NBP regulation. Economic research has found that the NBP led firms to install costly abatement technologies, and that regulated electricity generating units were especially likely to install highly effective "Selective Catalytic Reduction" abatement technology (Fowlie 2010). Engineering estimates suggest that the marginal abatement cost of $NO_x$ through this program is much larger than the marginal abatement cost of $NO_x$ from vehicles (Fowlie, Knittel, and Wolfram 2012). An integrated assessment model simulating costs and benefits of this market finds that the NBP as actually implemented was more cost-effective than an alternative design which recognized that damages vary across space because actual abatement costs exceeded ex ante expectations (Fowlie and Muller 2013).

Greenstone 2011). In contrast, the NBP provides quasi-experimental variation in emitted and ambient air pollution at the 5-month level (i.e., May 1st through September 30th); in the case of ozone, this is effectively annual variation because ozone is only considered a health risk during the higher concentration summer months. For these reasons, the paper is less subject to concerns about harvesting and is well suited to shed light on efforts to control $NO_x$ emissions and ozone concentrations. Additionally, recent research has emphasized the importance of using quasi-experimental variation to obtain reliable estimates of the relationship between human wellbeing and air pollution, and the NBP provides variation that is plausibly unrelated to other determinants of wellbeing (Dominici, Greenstone, and Sunstein 2014).

Third, we are unaware of other studies that demonstrate the impact of an emissions market on ambient pollution and human health with real world data. Most evaluations of emissions markets combine engineering models of emissions abatement, chemistry models of pollution transport, and epidemiological dose-response models (e.g., Muller and Mendelsohn 2009). The limitations of this approach are underlined by our failure to find consistent evidence of an impact of the NBP market on particulates air pollution, which the models (and the EPA) projected as the primary channel for any health benefits. In contrast, this paper's analysis is conducted with the most comprehensive data file ever compiled on emissions, pollution concentrations, defensive expenditures, and mortality rates.

The rest of this paper is organized as follows. Section II reviews ozone formation and the NBP. Section III presents a simple model of defensive investments. Section IV describes data sources and the analysis sample. Section V discusses the econometric models. Section VI reports the results and Section VII conducts a cost-benefit analysis of the NBP and develops a measure of willingness to pay for ozone reductions. Section VIII concludes.

## II. Ambient Pollution and the NBP Emission Market

*A. Ambient Pollution.* The Clean Air Act was designed to control ambient levels of ozone and five other pollutants that harm health. Ozone differs from the other pollutants in two important ways. First, polluters do not emit ozone directly. Instead, ambient ozone concentrations are governed by complex nonlinear photochemistry that depends on two chemicals precursors – nitrogen oxides ($NO_x$) and volatile organic compounds (VOCs) – and sunlight and heat. The market we study operates only in summer because winter ozone levels in the Eastern U.S. are low, and ozone spikes to high peaks on hot and sunny days.

Second, the health consequences of ozone are believed to occur from short-term exposure to high levels (Lippman 2009). Ozone regulation has targeted these peak exposures, rather than focusing on mean ozone levels. For example, the National Ambient Air Quality Standards for ozone primarily reflect the highest few readings of the year. Most epidemiological studies focus on very short-term effects, though some evidence suggests that medium- or long-run effects are larger (Jerrett et al. 2009). Hence, this market is most likely to affect health if it truncates the right tail of the ozone distribution.

*B. The $NO_x$ Budget Trading Program.* As detailed in Appendix IV, an initial version of the NBP operated in 1999-2002 and produced small declines in summer $NO_x$ emissions that are unlikely to confound our analysis of the 2003-2008 NBP (see Appendix IV). A more stringent version of the NBP then began in 2003 and operated until 2008.[8] This market included 2,500 electricity generating units and industrial boilers, though the 700 coal-fired electricity generating units in the market accounted for 95 percent of all NBP $NO_x$ emissions (USEPA 2009b).

The NBP market was implemented partially in 2003 and fully in 2004-5. The 2003-2008 emissions market originally aimed to cover the eight Northeast states plus Washington DC, plus 11 additional Eastern states. Litigation in the Midwest, however, delayed implementation in the

[8] In 2009, the Clean Air Interstate Rule (CAIR) replaced this market. In 2010, the EPA proposed a Transport Rule which would combine this $NO_x$ market with a market for $SO_2$ emissions. In July 2011, the EPA replaced this proposal with the Cross-State Air Pollution Rule, which regulates power plant emissions in 27 states with the goal of decreasing ambient ozone and particulate levels.

11 additional states until May 31, 2004.[9] Appendix Figure 1 shows the division of states by NBP participation status in the subsequent analysis.[10]

Accordingly, the EPA allocated about 150,000 tons of $NO_x$ allowances in 2003, 650,000 tons in 2004, and about 550,000 tons in each of the years 2005-2008.[11] Each state received a set of permits and chose how to distribute those permits to affected sources. Once permits were distributed, affected sources could buy and sell them through open markets. A single emissions cap affected the entire market region, though firms could bank allowances for any future year. Many firms banked allowances: In each year of the market, about 250,000 tons of allowances were saved unused for subsequent years (USEPA 2009a). At the end of each market season, each source had to give the EPA one allowance for each ton of $NO_x$ emitted. Seventy percent of units complied by using emissions controls (e.g., low $NO_x$ burners or selective catalytic reduction), and the remainder complied exclusively by holding emissions permits (USEPA 2009b). The mean resulting permit price in the emissions market was \$2,523 per ton of $NO_x$ (\$2015). This reflects the marginal abatement cost of the last unit of $NO_x$ abated, and we use it to develop an upper bound on the aggregate abatement cost of the NBP.

## III. Model of Willingness-to-Pay

We build upon the canonical Becker-Grossman health production function to highlight the role of defensive investments in the measurement of willingness-to-pay for clean air (Becker 1965; Grossman 1972). This model shows that accurate measurement of willingness-to-pay requires knowledge of both how pollution affects health outcomes such as mortality and how it affects defensive investments that maintain health but otherwise generate no utility, such as medications.

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[9] The 1999-2002 Ozone Transport Commission Market included Connecticut, Delaware, DC, Maryland, Massachusetts, New Jersey, New York, Pennsylvania, and Rhode Island. On May 1, 2003, the NBP emissions cap applied to the exact same set of states. On May 31, 2004, it also began applying to Alabama (excluding a southern region of the state), Illinois, Indiana, Kentucky, Michigan, North Carolina, Ohio, South Carolina, Tennessee, Virginia, and West Virginia. Missouri entered the market in 2007.

[10] The main results define all states in the NBP cap-and-trade region as treated, they exclude states that are adjacent to NBP states, and they define remaining states as comparison (non-treated). We exclude states adjacent to the NBP region from the main results because their treatment status is ambiguous (see Appendix IV for details).

[11] All tons in the paper refer to short tons and not metric tons.

Assume the sick days $s(d)$ which a person suffers depends on the dose $d$ of pollution she is exposed to. The effective dose $d(c,a)$ depends on the ambient concentration $c$ of the pollutant and on the defensive behavior $a$. Defensive behaviors can be taken before or after pollution is ingested—in the terminology of Graff-Zivin and Neidell (2013), defenses include both averting and mitigating activities. Substituting provides the following health production function:

(1) $s = s(c,a)$

People gain utility from consumption of a general good $X$ (whose price is normalized to 1), leisure $f$, and health. Budgets are constrained by non-labor income $I$, the wage rate $p_w$, available time $T$, and the price $p_a$ of defensive investments: $max_{X,f,a}u(X,f,s)$ s.t. $I + p_w(T - f - s) \geq X + p_a a$. Assuming an interior solution to the maximization problem, we can rearrange the total derivative of the health production function (1) to give the following expression for the partial effect of ambient pollution on sick days:

(2) $$\frac{\partial s}{\partial c} = \frac{ds}{dc} - \left(\frac{\partial s}{\partial a}\frac{\partial a^*}{\partial c}\right)$$

This expression is useful because it underscores that the partial derivative of sick days with respect to pollution is equal to the sum of the total derivative and the product of the partial derivative of sick days with respect to defensive behavior (assumed to have a negative sign) and the partial derivative of defensive behavior with respect to pollution (assumed to have a positive sign). In general, complete data on defensive behavior is unavailable, so most empirical investigations of pollution on health (see, e.g., Chay and Greenstone 2003) reveal $ds/dc$, rather than $\partial s/\partial a$. As equation (5) demonstrates, the total derivative is an underestimate of the desired partial derivative. Indeed, it is possible that virtually all of the response to a change in pollution comes through changes in defensive behavior and that there is little impact on health outcomes; in this case, an exclusive focus on the total derivative would lead to a substantial understatement of the health effect of pollution. The full impact therefore requires either estimation of $\partial s\partial da$, which is almost always infeasible, or of $ds/dc$ and $\partial a^*/\partial c$. We emphasize that defenses used both before and after pollution is ingested (i.e., averting and mitigating activities) are indistinguishable in the willingness-to-pay expression (2). From the view of social welfare, the distinction between them is not relevant.

To express the marginal willingness to pay for clean air $w_c$ in dollars, we manipulate the previous expressions to obtain the following decomposition:

$$(3) \qquad w_c = \left( p_w \frac{ds}{dc} \right) + \left( p_a \frac{\partial a^*}{\partial c} \right) - \left( \frac{\partial u / \partial s}{\lambda} \frac{ds}{dc} \right)$$

Expression (3) shows that the marginal willingness to pay for clean air includes three terms. The first is the effect of pollution on productive work time, valued at the wage rate. The third is the disutility of sickness, valued in dollars. This third component includes mortality. The second is the cost of defensive investments, valued at their market price. This second component is the aspect of willingness-to-pay that existing research has not measured. It is important to note that medications are not a complete measure of defensive investments against air pollution. The paper's primary empirical goal is to develop a measure of marginal willingness to pay that is based on $ds/dc$ and $\partial a^* \partial/c$.

Our setting has two important deviations from this neoclassical model: markups and moral hazard. Branded medications generally have low marginal cost and high markups that reflect intellectual property rights. Hence, it is natural to question whether changes in medication purchases amount to a transfer from consumers to drug firms, and not a social cost. In the short-run, this is indeed the case. However, pharmaceutical firms must invest socially valuable resources to develop medications that treat conditions exacerbated by air pollution. With lower levels of air pollution, fewer resources would be spent to develop these medications. Thus over the long run, there is a social benefit (see Finkelstein 2004) for a similar induced innovation process.

The second important deviation is that the marginal cost to the consumer is smaller than the price, by 80 percent in our data, because consumers with insurance generally pay a copayment or deductible for medications. We report medication cost results both using the full transacted price for medications (which is more accurate than the published or wholesale price) and using the copayment. The copayment may provide the best measure of a person's private willingness-to-pay for her own medications. Since an insurer must pay the remainder of the medication cost, the full cost of the medication may more accurately represent social-willingness-to-pay for the cost of the medications.

## IV. Data

This analysis has compiled an unprecedented set of data files to assess the impacts of the $NO_x$ Budget Program. Although market-based instruments are viewed as among the most important contributions of economics to environmental policy, to the best of our knowledge this study represents the first time any analysis has linked ex post health measurements directly to emissions and air quality measures in order to evaluate an emissions market.

*Medications.* We use confidential data on medication and hospital admissions from MarketScan. MarketScan contracts with large employers to obtain all insurance-related records for their employees and their dependents including children. The data report the purchase county, date, the medication's National Drug Code (NDC), and the money paid from consumer and insurer to the medication provider.

We use data from all persons in the 19 covered firms which appear in all years, 2001-2007, of MarketScan, which is the largest panel the data allow us to obtain with these firms. This extract includes over 22 million person-season year observations, and over 100 million separate medication purchases. Because the distribution of persons across counties is skewed, we report all values as rates per 1,000 people, and use generalized least squares (GLS) weights equal to the square root of the relevant MarketScan population. Because the other datasets become available in 1997 but medication data become available in 2001, for non-medication results we report parameter estimates both with data for the period 1997-2007 and for the period 2001-2007.

Medications are not linked to a single International Classification of Disease (ICD) code. In the subsequent analysis, we follow the convention in the pollution-health literature and treat respiratory and cardiovascular related episodes as most likely to be affected by air pollution. We define an NDC as respiratory if it satisfies any of three criteria: (1) if it is listed in the Third Treatment Guidelines for Asthma (NHLBI 2007); (2) in a recent New England Journal of Medicine guide to asthma treatment (Fanta 2009); or (3) in the standard industry publication for medication characteristics (PDR 2006) as indicated for asthma, emphysema, bronchitis, or chronic obstructive pulmonary disorder. We identify cardiovascular medications by their corresponding therapeutic group in Red Book (PDR 2006).[12]

This broad approach to identifying respiratory and cardiovascular drugs is the most appropriate we can discern. Nonetheless, because doctors prescribe medications to treat conditions for which the medications are not indicated, some of these medications were probably prescribed for non-respiratory and non-cardiovascular conditions. Moreover, it is also likely that medications prescribed for respiratory and cardiovascular conditions are not in this list. For example, the three sources mentioned above that we use to define respiratory medications have somewhat different categorization of which medications are respiratory. Internet searches for respiratory medications also find medications which can be used for respiratory conditions, but which are not listed as respiratory in any of the sources above.[13] Additionally, Red Book identifies a single therapeutic group for each National Drug Code. Since a medication may be used to treat multiple conditions, medications in non-cardiovascular therapeutic groups may also be used to treat cardiovascular conditions.

*Hospitalizations.* We count hospital admission costs as including all inpatient episodes plus all emergency outpatient episodes. When a hospital visit has several associated procedures each with its own ICD9 code, we take the mode procedure. Our measure of hospital costs includes all charges from the hospital to the insurer and patient.

[12] Red Book has no category for respiratory medications. Medication purchase rates are skewed and few county-season values equal zero, so the main tables report medication regressions in logs, with values of zero excluded from the regressions.

[13] For example, dexamethasone is not listed as respiratory in any of our sources, but medical websites like the Mayo Clinic's list it is as used to treat asthma along with many other conditions including inflammation, allergies, arthritis, blood or bone marrow problems, kidney problems, skin conditions, and multiple sclerosis. Similarly, isoflurane is not listed as a respiratory condition in our data, as it is primarily used for anesthesia, but many medical journal articles and other sources document its use for asthma.

*Mortality.* To measure mortality, we use restricted-access data on the universe of deaths in the 1997-2007 period. These Multiple Cause of Death files (MCOD) come from the National Center for Health Statistics (NCHS) and were accessed through an agreement between NCHS and the Census Research Data Centers. These files contain information on the county, cause of death, demographics, and date of each fatality.

*Pollution Emissions.* To measure pollution emissions, we extract daily totals of unit-level $NO_x$, $SO_2$, and $CO_2$ emissions for all states from the EPA's Clean Air Markets Division. The $NO_x$ emissions almost entirely come from CEMS and are quite accurate. Units which are part of the Acid Rain Program must report $NO_x$ emissions throughout the year, while units in the NBP must report $NO_x$ emissions only in the May 1 – September 30 period. Because we compare summer versus winter and East versus West, estimates in the paper use only data from Acid Rain Units. However, in the examined period, units in the NBP and not in the Acid Rain Program make a very small share of $NO_x$ emissions.

*Ambient Pollution.* We use a few criteria to select ambient pollution monitoring data from the EPA's detailed Air Quality System. Many EPA monitors operate for limited timespans and may change reporting frequency in response to pollution (Henderson 1996). The main analysis uses a fairly strenuous selection rule of limiting to monitors which have valid readings for at least 47 weeks in all years 1997-2007. Appendix Table 1 shows that we obtain similar results with a weaker monitor selection rule. For ozone, we focus on a concentration measure the EPA regulates: for each day, we calculate an "8-hour value" as the maximum rolling 8-hour mean within the day. We also calculate the number of days on which this 8-hour value was equal to or greater than 65 ppb, which is an indicator of high-ozone days.

*Weather.* We compiled daily maximum and minimum temperature, total daily precipitation, and dew point temperature data from records of the National Climate Data Center Summary of the Day files (File TD-3200). Appendix III explains the procedure chosen to ensure accurate and complete weather readings.

*Summary Statistics.* Table 1 shows the means, standard deviations, and county representation for the main variables in our analysis. Of the 2,539 counties in our preferred sample, medication and hospitalization data are available for 96 percent of these counties, which had a population of 261 million in 2004.[14] Ambient ozone data are only available for only 168 counties, but these counties are heavily populated and their 2004 population was 97 million. Data on particulates less than 2.5 micrometers ($PM_{2.5}$) are available in 298 counties (population 144 million) and data on particulates less than 10 micrometers ($PM_{10}$) are available for 39 counties (population of 26 million).

The summary statistics in Table 1 also provide a benchmark to measure the economic importance of medications and the emissions market. In summer, ozone averages 48 ppb. The 2010 proposed EPA air quality standard stipulated that a county could have no more than 3 days over a total of three years which exceed 60-70 ppb. Table 1 shows that during the sample period, 24 days every summer exceed 65 ppb in the typical county. On average during this time, the average person spent $378 per summer on medications, and about $600 on hospital admissions.

In unreported results, we also investigated potential unobserved variables in the observational associations between ozone and health. We divided all counties with ozone data into two sets—one set with mean summer ozone above the national median ("high ozone"), and another with mean summer ozone below the national median ("low ozone"). All ambient pollutant measures except carbon monoxide have significantly higher levels in the high-ozone counties. Temperature, precipitation, and dew point temperature are lower in high-ozone counties. The finding that so many of these observed county characteristics covary with ozone suggests that an observational association of ozone with health is likely to reflect the contributions of other unobserved variables and may explain the instability of the estimated health-ozone relationship that has plagued the previous literature. It is apparent that the estimation of the causal effect of $NO_x$ emissions and ozone on health and defensive expenditures

[14] While the U.S. has about 3,000 counties, our working sample is smaller since as discussed earlier, the main sample excludes several states adjacent to the NBP region since their treatment status is ambiguous.

requires a research design that isolates variation in $NO_x$ and ozone that is independent of potential confounders.

## V. Econometric Model

We use a differences-in-differences-in-differences (DDD) estimator to isolate the causal effects of the emissions market on pollution, defensive investments, and health, and use instrumental variables to measure the "structural" effect of $NO_x$ emissions and ozone on the same outcomes. The DDD estimator exploits three sources of variation in the emission and health data. First, we compare the years before and after the NBP's operation. Eight states plus Washington DC initiated this market in 2003, while 11 other states joined in 2004. This market did not operate before 2003. Second, 19 states plus Washington DC participated in the NBP while twenty-two other states did not participate and were not adjacent to a NBP state (see Appendix Figure 1). Third, the NBP market only operated during the summer, so we compare summer versus winter.

Specifically, we estimate the following model:

$$(7) \qquad Y_{cst} = \gamma_1 1(NBP \quad Operating)_{cst} + W'_{cst}\beta + \mu_{ct} + \eta_{st} + \nu_{cs} + \varepsilon_{cst}$$

Here, $c$ references county, $s$ indicates season, and $t$ denotes year. The year is divided into two seasons, summer and winter. Summer matches the NBP's operation period of May 1-September 30. The variables $Y_{cst}$ are pollution emissions, ambient pollution concentrations, medication costs, hospitalization costs, and mortality rates. Because the NBP market started partway in 2003, we define Post=0.5 in 2003 and Post=1.0 in 2004 through 2007. Appendix Tables 1, 2, and 4 show similar results for all natural alternatives to this definition. All regressions limit the sample to a balanced panel of county-season-years. Our main results cluster standard errors by state-season, but the Appendix reports alternative levels of clustering, with similar conclusions.

Since temperature has nonlinear effects on health, it is important to adjust for weather flexibly. The matrix of weather controls, $W_{cst}$, includes measures of precipitation, temperature, and dew point temperature (a measure of humidity). For temperature and humidity, we calculate 20 quantiles of the overall daily distribution.[15] For each county-season-year observation in the data, we then calculate the share of days that fall into each of the 20 quantiles.

To operationalize the DDD estimator, the specification includes all three sets of two-way fixed effects. The vector $\mu_{ct}$ is a complete set of county by year fixed effects, which account for all factors common to a county within a year (e.g., local economic activity and the quality of local health care providers). The season-by-year fixed effects, $\eta_{st}$, control for all factors common to a season and year: for example, they would adjust for the development of a new drug to treat asthma that was sold in NBP and non-NBP states. Finally, the county-by-season fixed effects, $v_{cs}$, allow for permanent differences in outcomes across county-by seasons. This specification estimates the difference in outcomes between a world with all $NO_x$ regulations including the NBP (including the Ozone Transport Commission market, RECLAIM, ozone nonattainment designations, and others) versus a world with all $NO_x$ regulations except the NBP. Other regulations did apply to $NO_x$ emissions from power plants in this period; for example, the Massachusetts State Implementation Plan adopted strict annual (though not summer-only) $NO_x$ emissions standards for power plants in 2001, which began applying between 2004 and 2008. Such policies help explain the downward trends observed in both Winter and in non-NBP states in Appendix Figure 2. Our identifying assumption is that such policies did not change differentially in NBP versus non-NBP states, in Winter versus Summer, over this period.

The parameter of interest is $\gamma_1$, associated with the variable $1(NBP\ Operating)_{cst}$. As noted earlier, this takes the value of 0.5 for all NBP states in 2003, when the market was operating in 9 of the 20 states, and a value of 1 in 2004 and all subsequent years in these states. The 2003 value is assigned to all NBP states, rather than just states which entered the market in 2003, because $NO_x$ and ozone travel far and emissions reductions in one NBP state affected ambient ozone in other NBP states. After adjustment for the fixed effects, $\gamma_1$ captures the variation in outcomes

[15] The lower quantiles of the precipitation distribution all equal zero, so for simplicity we specify the precipitation control as the mean level of precipitation in each county-year-summer.

specific to NBP states, relative to non-NBP states, in years when the NBP operated, relative to before its initiation, and in the summer, relative to the winter. This only leaves variation in the outcomes at the level at which the market operated.

Separate measures of the market's effect in each year provide additional useful information. Hence, for most outcomes, we also report the parameters $\alpha_{1997}\ldots\ \alpha_{2007}$ from the following model:

$$(8) \qquad Y_{cst} = \sum_{t=1997}^{2007} \alpha_t 1(NBP\ State\ and\ Summer)_{cs} + W_{cst}^{'}\beta + \mu_{ct} + \eta_{st} + \upsilon_{cs} + \varepsilon_{cst}$$

where *1(NBP State and Summer)*$_{cs}$=1 for all summer observations from NBP states, regardless of the year. We plot the $\alpha_t$'s in event study style figures to provide visual evidence on the validity of the conclusions from the estimation of equation (7).[16] Importantly, the event study style graphs provide an opportunity to assess whether there were pre-NPB trends in outcomes that were specific to NBP States after nonparametric adjustment for all county by year, season by year, and county by season factors. Appendix Figure 4 reports 20 separate event study graphs that cover all main outcomes in the paper.

Finally, we report on the results from the estimation of instrumental variables versions of

$$(9) \qquad Y_{cst} = \delta NOx_{cst} + W_{cst}^{'}\phi + \lambda_{ct} + \pi_{st} + \gamma_{cs} + \upsilon_{cst}$$

where the subscripts have the same meaning as in equations (7) and (8) and the equation includes the same set of fixed effects. Here, $Y_{cst}$ is restricted to measures of medication purchases and mortality rates. The key difference is that $NO_x$ emissions in county c, season s, and year t is an endogenous regressor and 1(NBP Operating)$_{cst}$ from equation (7) is used as an instrumental variable. We demonstrate below that there is a strong first-stage in that the instrumental variable predicts $NO_x$ emissions. The exclusion restriction is the other necessary condition for a valid instrumental variable and, conditional on the full set of two-way fixed effects, we believe that it

[16] The data on medication purchases and hospitalization begins in 2001, so for these outcomes, the event-study graphs are for the period 2001-2007.

is credible to assume that 1(NBP Operating)$_{cst}$ only affects medication purchases and mortality rates through $NO_x$ emissions.

The case for the validity of the exclusion restriction when ozone is the endogenous variable is plausible. However, it is less clear cut for two reasons: 1) the link between the NBP and ambient ozone is less direct since it is mediated by complex nonlinear photochemistry and this can make for a noisy relationship; and 2) air quality models show that atmospheric $NO_x$ can transform into particulates air pollution that is harmful to human health (Pandis and Seinfeld 2006). Nevertheless, there is a straightforward channel and we also report on versions of equation (9) where ambient ozone, instead of $NO_x$ emissions, is the endogenous variable, and 1(NBP Operating)$_{cst}$ is the instrumental variable. Reliable estimation of either, or both, versions of equation (9) would be of tremendous practical value for policy and, more broadly, so that this paper's results can be applied to other settings.

## VI. Results

*A. Emissions.* The $NO_x$ Budget Trading Program required affected units to reduce $NO_x$ emissions during the summer. Figure 2 (A) shows an event study graph measuring the difference between $NO_x$ emissions in the Eastern and Western U.S. and in summer versus winter, separately by year, with the year 2002 normalized to take the value zero. The value for 2001 is almost exactly equal to zero, which is consistent with a lack of pre-trends in $NO_x$ emissions. Figure 2 (A) shows that in the year 2003, when the NBP market began, $NO_x$ emissions fell by 0.2 thousand tons per county-season-year; and by the later years of the NBP, $NO_x$ emissions had fallen by a total of 0.3 to 0.4 thousand tons per county-season-year. Panel A of Table 2 reports estimates of several versions of equation (7) for pollution emissions measured at the county by season by year level. Column (1) includes county-by-season, season-by-year, and state-by-year fixed effects. Column (2) adds binned weather controls. Column (3) replaces the state-by-year fixed effects with county-by-year fixed effects, which causes the parameters of interest to be identified from comparisons of summer and winter emissions within a county by year. Column (4) restricts the sample to 2001-2007, which are the when medication and hospitalization data are available. Since emissions readings are totals rather than averages, the regressions are unweighted.

The entries in row 1 report the parameter estimate and standard error associated with the variable $1(NBP\ Operating)_{cst}$. The results suggest that the NBP market decreased $NO_x$ emissions in the average county by 330-430 tons. This corresponds to a total decrease of between 391,000 and 510,000 tons of $NO_x$ per summer.

It is informative to compare these statistics against other reports of the NBP's impacts on $NO_x$ emissions. The USEPA (2008) estimates that a combination of the NBP and its smaller predecessor, the $NO_x$ SIP Call, decreased ozone season $NO_x$ emissions by a larger amount, 750,000 tons. Their estimate comes from a time-series comparison of the years 2000 and 2007. Reconciling this with our smaller estimate is straightforward. A time-series comparison of the years 2002 and 2007 implies a smaller decrease of somewhat over 500,000 tons. Accounting for secular trends in emissions, which were present in both summer and winter seasons and in the NBP and non-NBP states (Appendix Figure 2), suggests an estimate within our range of 391,000 to 510,000 tons.[17]

We also measure whether the NBP market affected emissions of pollutants other than $NO_x$. Two economic reasons explain why the market might have affected emissions of such co-pollutants. If permits for $NO_x$ emissions cost enough that the market caused natural gas units to displace electricity generation from relatively dirty coal-fired units, then the market could have decreased emissions of pollutants other than $NO_x$. Second, complementarity or substitutability of $NO_x$ with other pollutants in electricity generation could lead units to change emissions of other pollutants. Rows 2 and 3 in Panel A of Table 2 indicate that NBP did not substantially affect $SO_2$ or $CO_2$ emissions. Our preferred estimates in column (3) are not statistically significant, though

[17] Our emissions totals are not numerically equal to those of at least the EPA's year 2008 NBP report for a few reasons: their report describes Missouri sources as regulated by an NBP in all years (whereas in reality those sources were only regulated in 2007; we exclude Missouri from our main analysis); we treat all of Alabama as in the NBP while they exclude some sources in southern Alabama; and we include only Acid Rain Units in the analysis since they have high-quality continuous emissions monitoring systems (CEMS) data for summer and winter and pre-NBP years, while the EPA's reports, which focus only on summer emissions, also include the fairly few NBP sources that are not in the Acid Rain Program.

some of the other estimates are. However, all of the estimates are economically small; for example, the point estimates in the preferred specifications in columns (3) and (4) are about 3% to 5% of the mean from years 2001-2, or a tenth of our proportional estimate for $NO_x$. Event study graphs in Appendix Figure 4(B) suggest quantitatively similar conclusions. The $SO_2$ graph, for example, suggests a decrease of 0.1 thousand tons per county-season, and the $CO_2$ graph suggests a decrease of approximately 20 thousand tons per county-season.

*B. Ambient Pollution.* Panel B in Table 2 reports on how the NBP affected ambient concentrations of ozone and the other pollutants that are most heavily regulated under the Clean Air Act. Columns (1) – (4) have identical specifications to those in Panel A, except that they are weighted by the number of pollution readings from the EPA's ambient air quality monitors in a given year by county. The column (5) estimates are from the same specification as in column (4), except that they are weighted by county population, which will be the relevant weight in the analysis of the impact of the NBP market on health outcomes (though defensive investments are weighted by the population in the MarketScan survey).

Rows 4 and 5 of in Panel B reveal large and precisely estimated effects of the emissions market on ground-level ozone concentrations (as measured by the maximum 8-hour value). The richest specifications in columns (3) - (5) indicate that the NBP decreased mean summer ozone by about 3 ppb (or 6% relative to the baseline average). Importantly, the NBP also decreased the number of high-ozone days (days where the 8-hour value equals or exceeds 65 ppb) by 8.0 to 9.6 days per summer (or 33%-40% of the baseline average). The corresponding event study figure for the 8-hour ozone reading (Appendix Figure 3 C) exhibits some evidence of differential pre-existing trends in summer ozone concentrations in NBP states. Accounting for these differences increases the magnitude of the NBP's estimated reduction on ozone concentrations, although these models are more demanding of the data and so the estimates are less precise, but remain significant at the conventional level.

Given the large effect of the NBP on the number of days with ozone equaling or exceeding 65 ppb, we also analyze the market's impact on the density function for daily ozone concentrations to explore where in the daily ozone distribution the NBP affected concentrations. Figure 2 (B) reports these results; the main finding is that the market reduced the number of summer days with relatively high-ozone concentrations (i.e. greater than 60 ppb) and increased the number of days with ozone concentrations less than 60 ppb. It is noteworthy that the EPA has experimented with daily ozone standards of 65, 75, and 85 ppb in recent years and that the identifying variation in ozone concentrations comes from this part of the distribution where there is great scientific and policy uncertainty.[18]

Rows 6-8 in Panel B of Table 2 test for impacts of NBP on carbon monoxide (CO), sulfur dioxide ($SO_2$), and nitrogen dioxide ($NO_2$). Appendix Figure 4(B) shows the corresponding event study graphs. CO emissions come primarily from transportation, so it is not surprising that the regressions fail to find evidence that the NBP affected CO emissions; the graphs bear this out. Further, there is no regression evidence of an impact on $SO_2$, though the event study graph has some evidence of pre-trends differences for this outcome. $NO_x$ is a standard term used to describe a mix of two compounds—nitric oxide (NO) and $NO_2$, a pollutant subject to its own regulations. Row 8 shows that the NBP market decreased ambient $NO_2$ levels by 6-7 percent, relative to the baseline, though $NO_2$ has limited or possibly no effect on health (Lippman 2009). The event study graph shows some decrease though is less clear than for ozone.

The impact of the NBP on particulates concentrations is of special interest because particulates can result from $NO_x$ emissions and are widely believed to be the most dangerous air pollutant for human health (Pope, Ezzati, and Dockery 2009; Chay and Greenstone 2003; Chen et al 2013). Further, before its implementation, the EPA estimated that 48-53 percent of the projected health benefits from the NBP would come through the channel of reduced particulates

[18] Appendix Figure 3 (A) shows the number of days with ozone in each of six bins in the years before the NBP program began. Appendix Figure 3 (B) shows event study graphs of the change in these counts due to the NBP. These graphs also show that the change in ozone was largely among days with 60-100ppb, which are exactly the set of days that regulation targets.

concentrations (USEPA 1998). The impact of emitted $NO_x$ on ambient particulate matter is theoretically ambiguous and depends on the level of other chemicals in the atmosphere (see Appendix I).

Rows 9 and 10 of Panel B in Table 2 empirically examine the impact of the NBP market on the concentrations of particles smaller than 10 micrometers ($PM_{10}$) and 2.5 micrometers ($PM_{2.5}$), both of which are small enough to be respirable. The $PM_{10}$ and $PM_{2.5}$ monitoring networks were just being erected in the late 1990s so to have meaningful samples it is necessary to focus on the 2001-2007 period as in columns (4) and (5). Column (4), where the equation is weighted by the number of monitor observations, provides limited evidence that the NBP affected particulate matter. Alternatively, when the equation is weighted by population, as is the case in the preferred defensive expenditures and health outcomes equations, the NBP is associated with a 7% reduction in $PM_{2.5}$. Because $PM_{2.5}$ is believed to create substantial health damages, however, the implications of this number for human health may be larger than this modest change in ambient concentrations might suggest. In the smaller sample of counties with $PM_{10}$ monitors, we fail to find evidence of a statistically significant change in $PM_{10}$. The row 9 and 10 results are inconclusive about whether the NBP affected particulates concentrations. The event study graphs in Appendix Figure 4(B) also show no clear evidence of a decrease in ambient particulates.

Overall, the large reduction in $NO_x$ emissions caused by the NBP market and the rest of the evidence in Table 2 is generally supportive of the premise that the effect of the NBP on health occurs primarily through its effect on ozone concentrations (see additional sensitivity analyses in Appendix V). Emissions of pollutants with important effects on health such as CO and $SO_2$ were unaffected by the NBP. However, the mixed estimates of the effect of the NBP on $PM_{2.5}$ (some statistically significant, some not) suggest that the subsequent 2SLS estimates of the effects of ozone on defensive expenditures and health outcomes derived from the variation in ozone induced by the NBP should be interpreted cautiously, because they may reflect the impact of ozone or particulates, or a combination of the two pollutants. We therefore focus more on the instrumental variables estimates of the effects of $NO_x$ emissions on medication purchases and mortality.

*C. Defensive Investments.* Table 3 statistically summarizes the reduced-form effect of the NBP market on log medication costs. The richest specification in columns (3) and (4) indicates that the NBP reduced total medication costs by 1.5 to 1.6 percent. The estimate is precise with the full set of controls and has similar magnitude but less precision with less detailed controls.[19] Finally, it is worth noting that the column (4) estimate is derived from the subsample of counties with ozone pollution monitors, which is used for the instrumental variables estimation below; this reduces the sample size from 30,926 to 2,338.

Figure 3 (A) shows the event study graph for log of respiratory and cardiovascular medication expenditures from the estimation of equation (8). The event study suggests that the NBP market decreased medication expenditures in these categories by nearly 2 percentage points. This impact was roughly constant and is marginally significant in individual years. Importantly, there is no evidence of meaningful differences in the trend in summertime medication purchases between NBP and non-NBP states in advance of the market's initiation. The picture is broadly similar though less precise for the smaller set of firms available over the period 2000-2007 (Appendix Figure 4B).

We also measure medication purchases separately by cause. As discussed above, the allocation of medications to causes is inexact—doctors can prescribe a medication for many purposes, and the MarketScan data do not identify the cause for which a specific medication was prescribed. The goal of this exercise is to test whether the decline in medication purchases was evident among respiratory and cardiovascular medications (although the imprecision of the assignment of causes to medications means that there are good reasons to expect an impact in

[19] County-by-year fixed effects add precision in these estimates. Because the medication data are from MarketScan and represent workers in the balanced panel of firms, county-by-year fixed effects address both local labor market shocks and firm- and factory-specific events like layoffs or mass hiring. Consistent with GLS providing an efficient response to heteroscedasticity, the unweighted estimate for log medication costs per capita is similar but less precise, at -0.011 (0.017).

other categories). The column (3) estimate in row 2 indicates that the NBP decreased expenditures on respiratory and cardiovascular medications by a statistically significant 2.1 percent. In the smaller sample of counties with ozone monitors in column (4), the point estimate is marginally different from zero and a similar point estimate to column (3). Expenditures on all other medications also declined in all specifications. In the richest specification of column (3), this decline is 1.4 percentage points and is modestly smaller than the decline for respiratory and cardiovascular medications. Event study graphs for non-cardiovascular and non-respiratory medications show much less evidence of a change than is apparent for cardiovascular and respiratory medications (Appendix Figure 4B).

An important question is the extent to which medications are a defense rather than just another health expense. Almost all of the previous literature reports on direct health outcomes (e.g., mortality rates, incidence of asthma attacks, and lung functioning). Following guidance from the medical literature (e.g., Fanta 2009), our paper's argument is that all of these health conditions are a function of ambient pollution and compensatory adaptations or defenses that include pharmaceutical purchases and a wide range of other costly actions. The share of willingness-to-pay accounted for by these defenses has essentially been unmeasured previously across a wide variety of settings; as a result, current measures of willingness to pay are incomplete and downward biased by an unknown magnitude. Appendix II discusses this question in detail, and while we argue that all medications are defensive, Appendix Table 2 reports results indicating that the NBP led to reductions in purchases of both short-acting acute and long-acting control respiratory medications; only the long-term control estimate is statistically significant.

*D. Hospital Visits and Mortality.*

*Hospital Visits*. Because we seek to compare defensive costs against direct health costs, we also measure how the market affected hospital visits and mortality. Due to the large number of county-year-season observations with zero hospitalization costs, we focus on the level rather than the log of per capita hospitalization costs.

Overall, our conclusion from these results is that we do not detect meaningful effects of the NBP on hospitalization costs and we do not pursue this outcome further (Appendix Table 3). We emphasize however that the MarketScan data exclude uninsured, Medicare, and Medicaid patients. These groups are included in some studies which find effects of ozone on hospitalization (Currie and Neidell 2005, Lleras-Muney 2010), and are believed to experience the largest impacts from high ambient ozone levels (ALA 2013). For these reasons, estimates of the effect of the NBP market on hospitalization (and potentially medications) could significantly understate population average effects.[20]

*Mortality*. In most analyses of air pollution, mortality accounts for the largest share of the regulatory benefits. The results in row 1 of Table 4 suggest that the NBP market decreased the all-cause, all-age summertime mortality rate by about 1.5 to 2.2 deaths per 100,000 population, depending on the sample, and would generally be judged to be statistically significant. The effect in the subsample of counties with ozone monitors is larger (see column 4), indicating a reduction of 5.2 deaths per 100,000 population.

Rows 2 through 4 of Table 4 divide the overall mortality rate by cause of death. Reading across row 2, it is apparent that 32% to 57% of the decline in overall mortality is concentrated among cardiovascular/respiratory deaths. Row 3 finds that the NBP also significantly decreased death from non-respiratory and non-cardiovascular causes. In most specifications the difference between the effects in rows 2 and rows 3 is smaller than one standard error, and the estimates are statistically indistinguishable.

Research on other pollutants finds that most of the health consequences of particulate matter are concentrated among respiratory and cardiovascular causes, although pathways for ozone are less well understood. The finding that the NBP affected respiratory and cardiovascular in addition to other causes is consistent with two hypotheses. One is that the NBP was correlated with unobserved shocks which affected mortality, and the second hypothesis is that the NBP

[20] At the same time, the MarketScan medication and hospitalizations data include insured groups that may be more prone than uninsured individuals to incur expenditures in response to health risks (Finkelstein et al. 2012).

itself caused these changes in mortality. Row 4 of Table 4 provides an important fact in support of the second hypothesis. Row 4 shows that the market had no effect on external (primarily accidental) deaths, which is a reassuring placebo test.

Panel C of Table 4 breaks the entire population into four age groups and separately estimates the effect of the NBP on each group's mortality rate using the full sample. The richest sample and specification in column (3) detects no statistically significant effect on the mortality of persons aged 74 and below, although the point estimates imply that the market prevented 424 deaths within this group. The largest impact on mortality occurs among people aged 75 and older. These results suggest that the NBP market prevented about 1,800 deaths each summer among people 75 and older. This finding is confirmed visually by the event study graph in Figure 3 (B), although the estimates from individual years are noisy, and by the age-specific analyses in Appendix Figure 4(B).

The age-group decomposition implies that the NBP prevented 2,237 summer deaths annually. About 80 percent of these were among people aged over 75. By contrast, the overall share of all summer deaths which occur among people aged over 75 is 55%, suggesting that the elderly disproportionately benefited from the NBP.

An important question that Table 4 leaves unanswered is the gain in life expectancy associated with these delayed fatalities. Indeed, the question of the magnitude of gains in life expectancy is unanswered in almost all of the air pollution and health literature because it is largely based on changes in mortality rates over relatively short periods of time (e.g., a few days or a week). The difficulty is that it is possible and perhaps likely that the relatively sick benefited and that their lifespans were extended only modestly, given their age. In the extreme, the NBP might merely have moved the date of these deaths to the winter months immediately following the market.[21]

[21] The paper's triple difference estimator compares summer and winter deaths within a year. If some of the deaths are displaced from summer to October-December of the same year, then the estimator will overstate the decline in mortality.

We explored two approaches to investigate the empirical relevance of this short-term ‘seasonal’ displacement hypothesis. First, we experimented with redefining each “year” to begin on May 1 of one calendar year and conclude on April 30 of the following calendar year. This redefined “year” compares each summertime season against the seven following months. Second, we estimated differences-in-differences regressions where each observation represents a calendar year (as opposed to a calendar-season-year), and where we measure the change in mortality rates by NBP status pre vs. post. We also combined these two approaches to estimate differences-in-differences models with the restructured year.

These approaches do not provide strong support for the short-term displacement hypothesis. In most cases, the estimated effect of the market on mortality was negative and had similar magnitude to the models reported in the paper, but these estimates were imprecise and we could not reject the null hypothesis that the NBP had no long-run impact on mortality. Overall, we conclude that this research design lacks power to measure the effect of ozone on life expectancy beyond the five month length of the NBP’s summer season. Nevertheless, this paper’s focus on the summertime mortality rate is an advance from the previous literature that has primarily estimated how ozone affects same-day or same-week mortality rates.[22]

*E. Instrumental Variables (IV).* The preceding sections measure the reduced-form effects of the NBP market on pollution, defenses, and health. We now turn to an IV approach to measuring the effect of $NO_x$ emissions and ozone on defensive expenditures and mortality rates. The interpretation of the IV estimates of the effect of $NO_x$ generated by the NBP market variation as causal is straightforward: $NO_x$ is a pollutant controlled by regulation, and the estimated effects on health and defenses are a direct result of the quasi-experimental change in $NO_x$ emissions. Table 2 showed that changes in $NO_x$ are the primary channel for the large changes in ambient ozone concentrations. However, changes in $NO_x$ – depending on the model specification -- also

[22] Currie and Neidell (2005) is an exception since they estimate monthly and quarterly mortality regressions.

can lead to changes in other ambient pollutants, including $PM_{2.5}$. Thus we underscore that instrumental variable estimates of the effect of ozone should be interpreted more cautiously.

We report IV estimates for the effect of $NO_x$ on health for two different geographic samples—all counties and the 24% of NBP counties with positive $NO_x$ emissions in summer 2002 that account for 44% of these states' population. We refer to the second sample as, "Counties with $NO_x$ Emissions." Most counties lack power plants that produce $NO_x$ emissions so the NBP could not have affected emissions in these counties, except by deterring entry; relatedly, the first-stage estimate of the NBP's effect on $NO_x$ is more powerful when excluding these zeros. We emphasize estimates from the second sample, since it is ex ante expected to have more statistical power. Of course we can only estimate IV regressions for ozone using the counties with ozone monitors.

The first row of Panel A of Table 5 reports fixed effects estimates of the association between $NO_x$ emissions and medication purchases (columns 1 - 2) and between measures of the all-age mortality rate (columns 3 – 4). Rows 2 and 3 repeat the exercise for two different measures of ambient ozone. The estimates are from separate regressions of the outcome on alternative measures of $NO_x$ emissions (or ozone concentrations) and are adjusted for county-by-season fixed effects, county-by-year fixed effects, season-by-year fixed effects, detailed weather controls, and each observation represents a county-year-season as in the prior analysis. Most of these estimates are statistically insignificant, and exhibit sign and magnitude variability (including perversely signed coefficients on mortality), suggesting little evidence of systematic effects on medication purchases or mortality rates.

Panel B reports on the two-stage least squares (2SLS) or instrumental variables (IV) estimates that are adjusted for the same controls as in the fixed effects specifications but the endogenous variables (i.e., $NO_x$ emissions in thousands of tons, average 8-hour ozone concentration, and the number of days equaling or exceeding 65 ppb) are instrumented using the quasi-experimental variation generated by the NBP market. That is, we use the variable $1(NBP\ Operating)_{cst}$ as an instrument for $NO_x$ emissions, and for ambient ozone concentrations.

The entries indicate a strong relationship between $NO_x$ emissions and medication purchases. For example, the estimates based on the sample of counties with positive $NO_x$ emissions imply that a 10 percentage point decline in $NO_x$ emissions relative to the Table 1 mean of 0.52 leads to a 0.07 percentage point reduction in spending on all medications. The estimates including all counties are larger, at 0.14 percentage points, though less precise. Both estimates are substantially larger in magnitude than the analogous OLS ones in Panel A, which is consistent with the possibilities that the OLS estimates are plagued by substantial confounding and that $NO_x$ emissions are measured with error. The ozone entries imply that a 10 percentage point decline in the average 8 hour ozone measure and a 10 percentage point decline in days with ozone concentrations exceeding 65 ppb reduces all medication spending by 2.8 and 0.37 percentage points, respectively (both statistically significant).

The IV mortality estimates in columns (4) also imply large mortality effects of $NO_x$ emissions and ozone concentrations.[23] The estimates based on counties with $NO_x$ emissions suggest that a 1 million ton increase in $NO_x$ emissions leads to 5 additional summertime deaths per 100,000 people, or that a 10 percentage point decline in $NO_x$ leads to a reduction in the mortality rate of 0.07 percentage points. The estimates for all counties are the same magnitude though somewhat less precise. The estimates also indicate that a 1 ppb increase in the 8 hour ozone concentration or 1 additional day with a concentration exceeding 65 ppb lead to 2.62 and 0.86 additional summertime deaths per 100,000 people, respectively; correspondingly, a 10 percentage point increase in 8-hour ozone or in days with ozone concentrations exceeding 65 ppb

[23] The first-stage regression for column (3) of Table 5 corresponds to column (5) of Table 2. For the other columns of Table 5, the first stage regressions are as follows: for column (1), -0.704 (0.303); for column (2a), -1.500 (0.518); for column (2b), -0.107 (0.051); for column (4a), -0.876 (0.274); for column (4b), -0.237 (0.095). The sample and weighting (MarketScan versus total population) varies across these regressions. Appendix Table 7 reports 2SLS estimates separately for respiratory and cardiovascular causes (Panel A), and for non-respiratory and non-cardiovascular causes (Panel B). These 2SLS estimates by cause are somewhat less precise than the all-cause estimates in Table 5, though they still show that a substantial portion of the health consequences of $NO_x$ emissions or ozone exposure come through respiratory and cardiovascular causes.

leads to a 3 or an 0.46 percentage point increase in summertime deaths per 100,000 people, respectively.[24]

To the best of our knowledge, this is the first paper to develop plausibly causal estimates of the relationships between $NO_x$ emissions with health and defensive investments.[25] Further if it is appropriate to interpret the IV ozone estimates as causal, they would substantially alter our understanding of the welfare consequence of exposure to ozone. For example, the most prominent ozone-mortality study (Bell et al. 2004) finds an elasticity of weekly ozone with respect to daily mortality rates that is smaller than the elasticity implied by Table 5.

## VII. Calculating Welfare Impacts

This paper's results allow us to conduct a simple cost-benefit analysis for the entire NBP, with the caveat that data restrictions prevent us from measuring all health outcomes and defensive expenditures. The estimates in Table 2 imply that the NBP market decreased $NO_x$ emissions by 427,000 tons per summer on average and the average cost of a $NO_x$ permit was $2,523/ton.[26] The permit price should reflect an upper bound on abatement costs per ton, because firms should only use abatement technologies that cost less than the permit price. Thus, an upper bound estimate is that the market caused firms to spend $1,076 million (2015$) annually to abate $NO_x$ (Table 6, Column 4). Defining 2003 to have half a year of typical abatement costs, we obtain an upper bound on 2003-2007 total abatement costs of $4.8 billion (=1,076 x 4.5).

We now turn to estimating the NBP's social benefits. As we discussed above, it may seem natural to assume that a change in pharmaceutical purchases are simply a transfer from

[24] The values in Table 5 all include the years 2001-2007. For mortality but not medications we can also estimate results including years 1997-2007, and obtain generally similar results. With this longer time period, the 2SLS estimate for the mortality effect of $NO_x$ emissions in all counties is 5.18 (3.50), or 2.67 (1.89) in counties with $NO_x$ emissions. The 2SLS estimates for 8-hour ozone and ozone days above 65 ppb are 1.61 (0.76) and 0.48 (0.23), respectively.

[25] Holland et al. (2016) apply the Muller-Mendelsohn model to simulate the consequences of emissions from Volkswagen vehicles. They do not estimate regressions directly linking observed $NO_x$ emissions to observed outcomes like health.

[26] The effect on $NO_x$ emissions is calculated by multiplying the estimated impact of NPB on $NO_x$ emissions (-0.36) in thousand tons from Table 2, Column (3), by the number of counties in the NBP (1,185).

consumers to pharmaceutical firms and thus have zero social cost. However, reductions in air pollution concentrations decrease the demand for medications that protect individuals from air pollution. Dynamically, this decline in demand will reduce the resources used to develop these medication types and will allow these resources to be applied to more productive uses. We are unaware of an empirically validated approach to socially valuing this reduction in drug purchases but believe that it is defensible to assume that it is valued at their full cost, especially over long time horizons; Table 6 adopts this assumption, and also reports the total change in copayments.

Column (1) of Table 6 Panel B reports the average annual reduction in medication expenditures, as well as the sum over the NBP's life. Specifically, we take the estimated 1.5% reduction in medication purchases from the regression result in column (3) and row 1 of Table 3 and multiply that by the annual mean medication purchases. This calculation suggests that the NBP market led to a decrease in medication expenditures of $820 million per year or $3.7 billion when summed over the 4.5 years that the NBP operated. It is unclear whether this extrapolation from the MarketScan population under- or over-states the effect on the full population.

The Table 4 mortality estimates imply that the market prevented about 2,200 deaths per summer. The value of a statistical life (VSL) determines the monetary value assigned to these deaths. To provide one approach to monetization, we use Ashenfelter and Greenstone's (2004) upper bound VSL of $2.27 million (2015$) for a prime age person and Murphy and Topel's (2006) method to develop estimates of the VSL for each age group in our analysis. This adjustment is especially consequential in this setting where the avoided fatalities are largely among individuals 75 and over. The implied VSLs are as follows: $2.3 million (infants), $1.78 million (ages 1-64), $0.7 million (ages 65-74), and $0.3 million (ages 75+). The application of this approach implies that the value of the mortality avoided by the NBP is $1.1 billion per year, or $4.8 billion in the period 2003-2007 (Table 6 Panel B).[27]

[27] We thank Kevin Murphy and Bob Topel for sharing the data underlying Figure 3 of their paper. The VSL used here is lower than the $8.7 million VSL ($2015) used by the EPA, which is not age-adjusted. Our primary goal is not to endorse a specific VSL value, but to demonstrate the results that come from one choice of VSL and age-

The entries in Panels A and B of Table 6 provide the basis for a comparison of the costs and benefits. The upper bound on the NBP's aggregate abatement costs is \$4.8 billion, but by themselves the value of the reduced medication purchases of \$3.7 billion nearly equals these costs. At least in this context, defensive investments are economically important. Once the value of the reduced rates of mortality is added in, the benefits of the market are nearly twice as large as the upper-bound of its abatement costs (i.e., \$8.5 billion in benefits and \$4.8 billion in costs). Thus the NBP's social benefits easily exceeded its abatement costs.

We consider two alternatives to the benefit-cost analysis in column 5 Table 6. An alternative measure of medication costs is copayments. The log change in copayments is almost identical to the log change in total medication expenditures (see Appendix Table 2). Copayments represent 21 percent of the total payment for medications (Table 1). Total copayment savings per year are \$150 million, or \$676 million over the 2003-2007 NBP. A second alternative is to calculate the upper-bound on the NBP market cost using other measures of the NBP allowance price. An alternative estimate of market costs of \$3,000 per ton implies an upper bound cost of \$1.3 billion per year, or \$5.8 billion total over the 2003-2007 period.[28] Under both alternatives, medication costs represent an economically important proportion of the benefits and costs of the NBP market, and excluding medication expenditures (which are only one component of defensive investments) would substantially understate the market's benefits. The magnitude of the understatement varies with these assumptions.

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adjustment. Using the \$8.7 million VSL rather than the \$2.27 million VSL implies that the mortality benefits of NBP were larger: \$5.3 billion per year or \$23.9 billion for the 2003-2007 total.

[28] This value slightly exceeds the average allowance price in the years 2002-2006. When the market opened in May 2003, allowance prices of different vintages ranged from \$2250 to \$5000 per ton. Prices of all vintages fell rapidly during the first year to below \$3000. In the following years 2004-2006, allowance prices were fairly stable at between \$2000 and \$3000 per ton. In 2007 all allowance prices declined to \$1000 per ton.

Estimates of willingness to pay for reductions in $NO_x$ emissions have considerable policy relevance since $NO_x$ is the pollutant that policymakers can regulate directly, whereas ozone is only formed through complicated chemical reactions involving other pollutants. Table 6's Panel C reports on estimated willingness to pay (WTP) for a reduction of one million tons of NOx emissions and its component parts.[29] Based on estimates from counties with $NO_x$ emissions, each 1 million ton decrease in summertime $NO_x$ emissions in the NBP states annually saves about $0.6 billion in medication expenditures and roughly 8,000 premature summertime deaths, with an estimated value of $3.9 billion in mortality benefits (Table 6, Panel C); the total WTP is thus about $4.5 billion. The alternative estimated WTP of $4.8 billion, based on estimates from the full sample of counties, is qualitatively identical. These figures are underestimates if other categories of well-being or defensive expenditures respond to changes in $NO_x$ emissions.

Table 6 also reports estimates of willingness to pay for a reduction in ozone, but they must be interpreted cautiously due to uncertainty about the validity of the exclusion restriction.[30] The IV ozone results suggest that each 1 ppb decrease in the mean 8-hour summer ozone concentration in the NBP states is worth approximately $1.9 billion in social benefits annually. Similarly, one fewer day per summer in the NBP states with an ozone concentration exceeding 65 ppb would yield roughly $600 million of benefits annually (Table 6, Panel D).

## VIII. Conclusions

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[29] The value in column (2) comes from multiplying together the IV estimates of the effect of NOx emissions on log medication costs (Table 5), the mean medication expenditure per person-season (Table 1), and the mean population in the NBP states in the years 2003-2007 (136 million). The value in column (2) comes from multiplying the IV estimates of the effects of NOx emissions on the mortality rate (Table 5) by the total population in the NBP states. The value in column (3) comes from by taking the mean willingness to pay per death prevented from Panel B of Table 6, and multiplying it by the change in number of deaths from column (2) of Table 6.

[30] The approach for calculating the ozone benefits in Panel D of Table 6 is similar to the methodology for calculating the $NO_x$ benefits in Panel C of Table 6, and is described above. It is worth noting that estimates of the benefits of the NBP, $NO_x$ emissions, and ozone for mortality and medications do not have identical samples since not all counties have ozone monitors, and that medications data are available only beginning in 2001.

Theoretical models make clear that willingness to pay (WTP) for well-being in a variety of contexts is a function of factors that enter the utility function directly (e.g., the probability of mortality, school quality, etc.) and the costly investments that help to determine these factors. One approach to developing measures of WTP is to find a single market that captures individuals' full valuation, as can be the case with property markets under some assumptions (see, e.g., Chay and Greenstone 2005; Greenstone and Gallagher 2008). All too frequently though, the data and/or a compelling research design for the key market outcomes are unavailable, making it necessary to develop measures of WTP by summing its components.

However, across a wide variety of applied literatures, the empirical evidence on WTP has almost exclusively focused on the factors that enter the utility function directly. The resulting measures of willingness to pay are thus generally underestimated and the extent of this underestimation is unknown. This paper has demonstrated that defensive expenditures are an important part of willingness to pay for air quality. Indeed in the context of the $NO_x$ Budget Program, the improvement in air quality generates reductions in medication purchases that are close to an upper bound estimate of the abatement cost. A fruitful area for research is to explore whether individuals' compensatory behavior and resulting defensive investments account for such a large fraction of willingness to pay in other settings.

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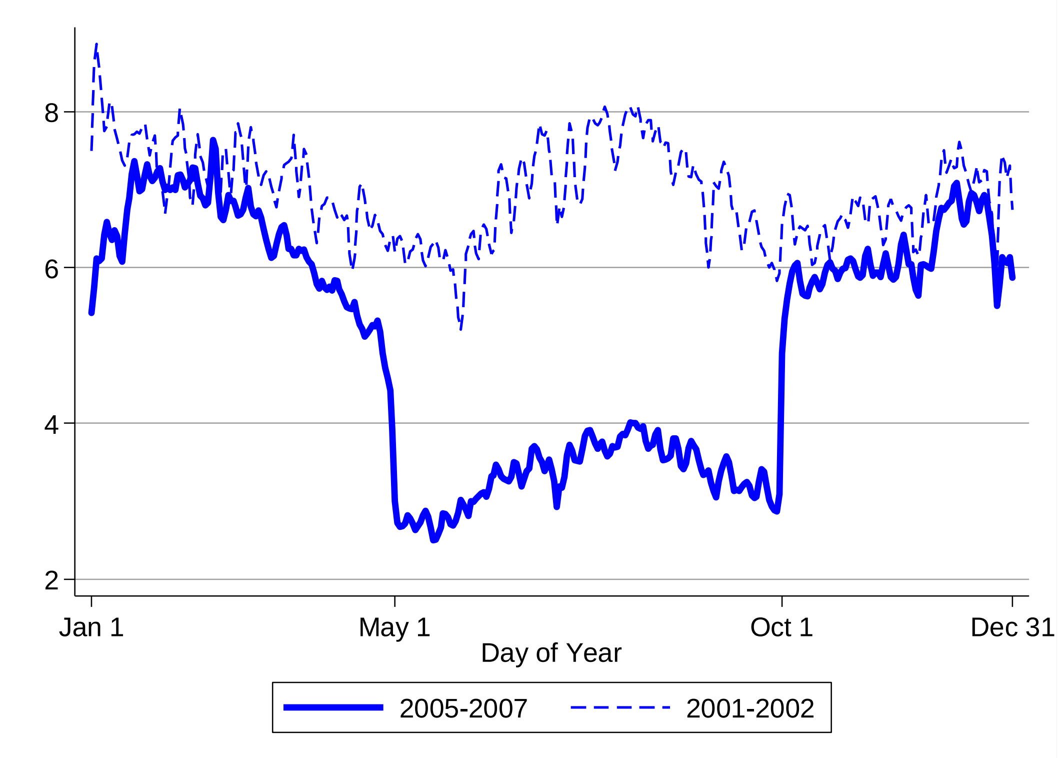
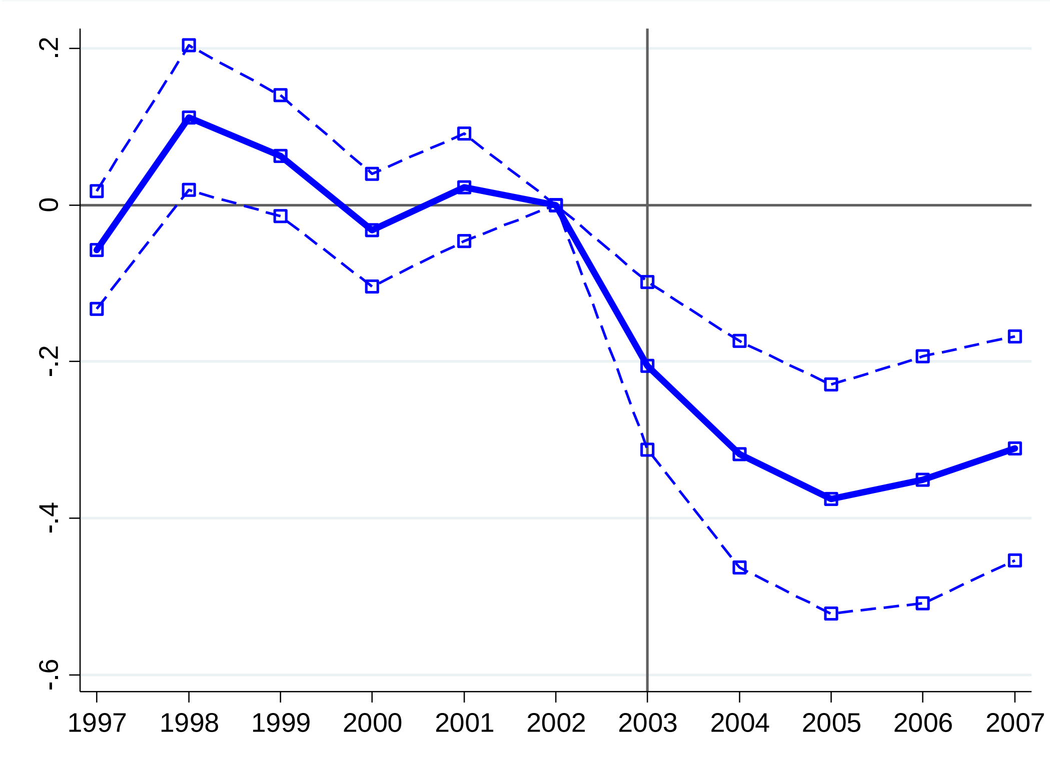
**Figure 1. Total Daily $NO_X$ Emissions in the NBP-Participating States**

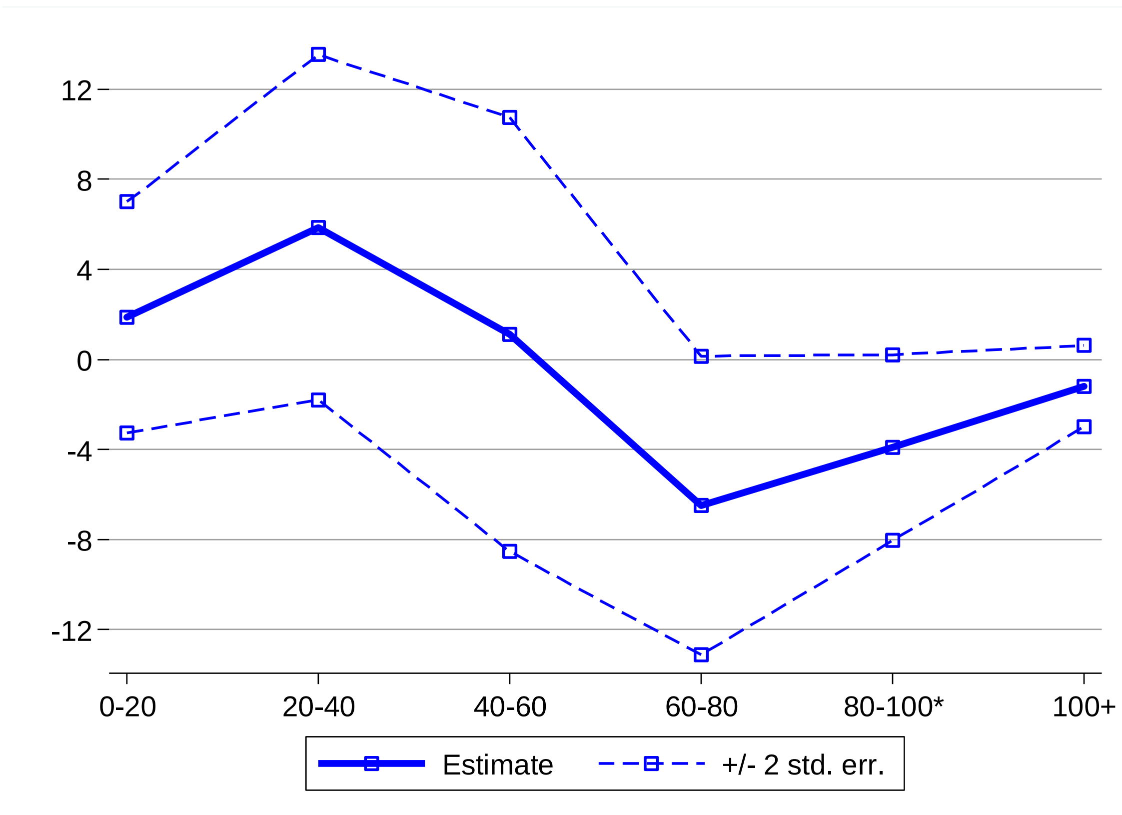



**Notes:** Figure 1 shows average total daily $NO_x$ emissions in the NBP participating states in 2001-02 and 2005-07. These estimates are obtained an OLS regression of $NO_x$ emissions on 6 day-of-week indicators and a constant. The values in the graph equal the constant plus the regression residuals, so that the graph depicts fitted values for the reference category (Wednesday). Total daily NOx emissions on Y-axis are measured in thousands of tons. The sample includes emissions from all the Acid Rain Units. NBP participating states include: Alabama, Connecticut, Delaware, District of Columbia, Illinois, Indiana, Kentucky, Maryland, Massachusetts, Michigan, Missouri, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Rhode Island, South Carolina, Tennessee, Virginia, and West Virginia. The NBP operated only in Northeastern states on May 1 of 2003, and expanded to the other states on May 31 of 2004. See the text for more details.

**Figure 2. NBP Market Impact on $NO_x$ Emissions and Ambient Ozone Pollution**
**(A) Event Study for $NO_x$ Emissions, 1997-2007**



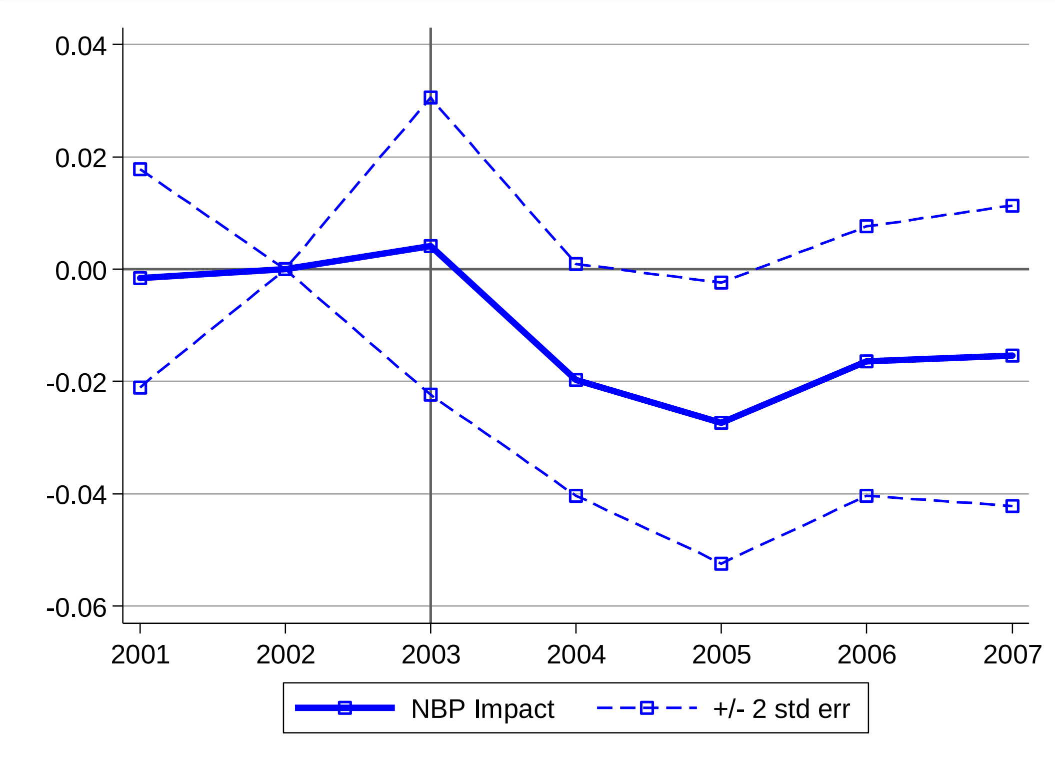

**(B) NBP Market Impact on the Number of Summer Days in Six Ozone Bins**



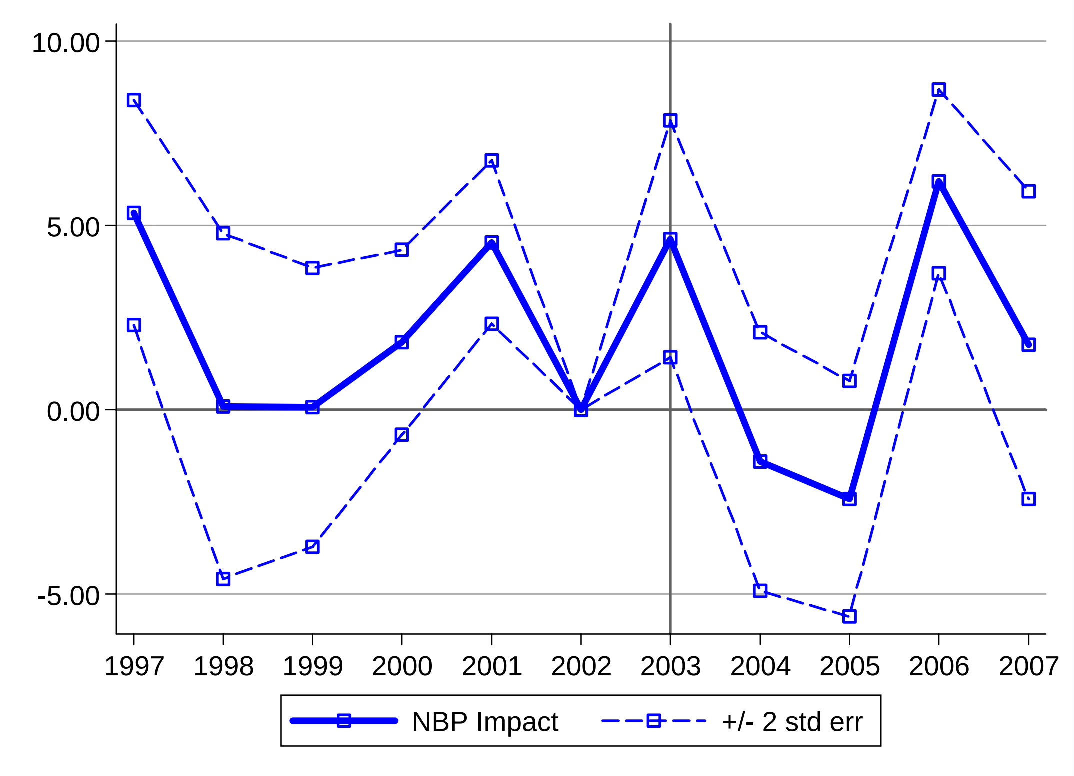
**Notes:** The estimates in Figure 2 Panel (A) are from an event study regression for $NO_x$ emissions (measured in thousand of tons and observed at the county*year*season) where the estimates for year 2002 are restricted to have a value of 0. The regression includes detailed weather controls and a full set of county*year, season*year, and county*season fixed effects. The standard errors underlying the 95% confidence intervals (dashed lines) are clustered at the state-season level. In Panel (B), the bins represent to 6 categories of ozone 8-hour values, corresponding to the number of days per summer where the ozone 8-hour value is 0-20 ppb, 20-40 ppb, 40-60 ppb, etc. The ozone 8-hour value is measured as the maximum rolling 8-hour mean of hourly values within each day. The coefficients reported in Figure 2 Panel (B) are from a separate regression for each bin that includes detailed weather controls and a full set of county*year, season*year, and county*season fixed effects. The regression is weighted by the number of ozone readings in each county*season*year. The standard errors underlying the 95% confidence intervals (dashed lines) are clustered at the state-season level. The asterisk on the 80-100 category denotes the nonattainment air quality standard during the NBP years (85 ppb).

**Figure 3. Impact of the NBP Market on Defensive Expenditures Health Outcomes**

**(A) Event Study for Log Respiratory + Cardiovascular Medication Costs Per Capita ($2015)**




**(B) Event Study for Respiratory + Cardiovascular Mortality Rate**



**Notes:** The estimates in Figure 3 Panel (A) are from an event study regression for log medication costs per capita (respiratory and cardiovascular medications only), and the estimates in Figure 3 Panel (B) are from an event study regression for mortality rates per 100,000 population (respiratory and cardiovascular causes only). In both regressions, the estimates for year 2002 are restricted to have a value of 0. The regressions include detailed weather controls and a full set of county*year, season*year, and county*season fixed effects, and are weighted by county population (Panel (A) uses MarketScan population and Panel (B) uses total population). The standard errors underlying the 95% confidence intervals (dashed lines) are clustered at the state-season level. Medication costs are in 2015 dollars, deflated using the BLS CPI for urban consumers.

**Table 1. Mean Summer Values of the Pollution, Weather, and Health Variables, 2001-2007**

| | Counties With Data (1) | Mean (2) | s.d. (3) |
|---|---|---|---|
| Pollution Emissions (000's of Tons/Summer) | | | |
| $NO_x$ Emissions | 2,539 | 0.52 | (1.99) |
| $SO_2$ Emissions | 2,539 | 1.50 | (6.52) |
| $CO_2$ Emissions | 2,539 | 384 | (1,299) |
| Air Quality (Ambient Pollution) | | | |
| Ozone 8-Hour Value (ppb) | 168 | 48.06 | (9.28) |
| Ozone Days $\geq$65 (ppb) | 168 | 23.60 | (22.64) |
| $NO_2$ (ppb) | 110 | 11.45 | (5.39) |
| CO (ppm) | 125 | 0.44 | (0.24) |
| $PM_{2.5}$ ($\mu g/m^3$) | 298 | 13.33 | (4.19) |
| $PM_{10}$ ($\mu g/m^3$) | 39 | 27.28 | (6.26) |
| $SO_2$ (ppb) | 150 | 3.26 | (2.27) |
| Weather | | | |
| Temperature (ºF) | 2,539 | 70.59 | (5.79) |
| Precipitation (1/100") | 2,539 | 11.46 | (5.37) |
| Dew Point Temp. (ºF) | 2,539 | 57.77 | (7.91) |
| Medication Costs ($ Per Person) | | | |
| All | 2,435 | 377.56 | (338.66) |
| Copayment | 2,435 | 79.10 | (59.21) |
| Respiratory or Cardio. | 2,435 | 122.67 | (131.87) |
| Hospitalizations ($ Per Person) | | | |
| All | 2,435 | 593.93 | (2,501.71) |
| Respiratory or Cardio. | 2,435 | 120.57 | (923.57) |
| Mortality (Deaths Per 100,000 People) | | | |
| All | 2,539 | 402.42 | (121.32) |
| Respiratory or Cardio. | 2,539 | 180.80 | (69.93) |

**Notes:** Medication and hospitalization costs are reported in 2015 dollars and deflated using the US CPI for urban consumers. Emissions, medications, and deaths are totals per summer. Ambient pollution and weather are mean summer values. Means are across counties (i.e., not weighted). All data is for the period 2001-2007.

## Table 2. Effect of the NBP Market on Emitted and Ambient Pollution

| | (1) | (2) | (3) | (4) | (5) |
|---|---|---|---|---|---|
| **A. Pollution Emissions (000's of Tons per Summer)** | | | | | |
| 1. $NO_x$ | -0.36*** | -0.37*** | -0.36*** | -0.33*** | -0.43*** |
| | (0.05) | (0.06) | (0.07) | (0.07) | (0.11) |
| 2. $SO_2$ | -0.08** | -0.12* | -0.07 | -0.07** | -0.05 |
| | (0.04) | (0.07) | (0.05) | (0.03) | (0.13) |
| 3. $CO_2$ | -2.66 | -20.57 | -4.04 | -12.43* | -79.96 |
| | (4.33) | (15.87) | (5.89) | (6.59) | (56.94) |
| **B. Air Quality (Ambient Pollution)** | | | | | |
| 4. Ozone 8-Hour Value (ppb) | -2.91*** | -3.74*** | -2.76*** | -3.06*** | -3.15*** |
| | (0.77) | (1.20) | (0.73) | (0.50) | (0.49) |
| 5. Ozone Days ≥ 65 (ppb) | -7.40*** | -8.53*** | -7.96** | -8.95*** | -9.64*** |
| | (2.50) | (2.76) | (3.03) | (2.61) | (2.32) |
| 6. CO: Carbon Monoxide (ppm) | -0.05** | -0.01 | -0.04 | -0.02 | -0.01 |
| | (0.02) | (0.03) | (0.03) | (0.02) | (0.03) |
| 7. $SO_2$: Sulfur Dioxide (ppb) | 0.16 | 0.34 | 0.14 | 0.13 | 0.13 |
| | (0.12) | (0.24) | (0.20) | (0.18) | (0.14) |
| 8. $NO_2$: Nitrogen Dioxide (ppb) | -1.13*** | -0.92 | -1.09*** | -0.92** | -1.23** |
| | (0.21) | (0.75) | (0.35) | (0.35) | (0.49) |
| 9. $PM_{2.5}$: Particulates Less than 2.5 Micrometers | --- | --- | --- | -0.31 | -0.93*** |
| (µg/m3) | --- | --- | --- | (0.31) | (0.27) |
| 10. $PM_{10}$: Particulates Less than 10 Micrometers | --- | --- | --- | -1.18 | -0.82 |
| (µg/m3) | --- | --- | --- | (0.87) | (1.05) |
| County-by-Season FE | x | x | x | x | x |
| Summer-by-Year FE | x | x | x | x | x |
| State-by-Year FE | x | x | | | |
| County-by-Year FE | | | x | x | x |
| Detailed Weather Controls | | x | x | x | x |
| Data Begins in 2001 | | | | x | x |
| Weighted by Emission/Pollution Monitors (B. only) | x | x | x | x | |
| Weighted by Population | | | | | x |

**Notes:**. The entries in Table 2 are the coefficient estimates from the DDD estimator described in equation (7). Each coefficient is from a separate regression that includes a full set of county*year, season*year, and county*season fixed effects. Additional control variables are listed in the text. The reported standard errors are clustered at the state-season level. Emitted pollutant variables (Panel A) are measured in thousand of tons and ambient pollutant variables (Panel B) are mean values. Unless otherwise noted, the sample period begins in 1997. Ambient pollution regressions (Panel B) are GLS weighted by square root of number of underlying pollution readings unless otherwise noted. For emissions, the number of observations is 55,858 for emissions in columns (1) to (3) and 35,546 for column (4). For ambient pollution, the number of observations for each pollutant based on 1997-2007 sample (2001-2007 sample for PM) is 3,124 (Ozone); 2,244 (CO); 4,172 ($PM_{2.5}$); 546 ($PM_{10}$); 2,684 ($SO_2$); 1,782 ($NO_2$). Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***). For emissions, share of population covered is 100%. For ambient pollution, share of population covered is 28-40 percent for ozone, CO, $SO_2$, and $NO_2$; 55 percent for $PM_{2.5}$, and 10 percent for $PM_{10}$.

## Table 3. Effect of the NBP Market on Log Medication Costs Per Capita

| | (1) | (2) | (3) | (4) |
|---|---|---|---|---|
| **A. All Medications** | | | | |
| 1. All Medications | -0.008 | -0.023 | -0.015** | -0.016*** |
| | (0.010) | (0.020) | (0.006) | (0.006) |
| MarketScan as Share of Total Population | 0.007 | 0.007 | 0.007 | 0.002 |
| Counties as Share of Total Population | 0.994 | 0.994 | 0.994 | 0.370 |
| **B. Specific Types of Medications** | | | | |
| 2. Respiratory or Cardiovascular | -0.009 | -0.022 | -0.021*** | -0.017* |
| | (0.013) | (0.023) | (0.008) | (0.010) |
| 3. Non-Respiratory and Non-Cardiovascular | -0.009 | -0.025 | -0.014** | -0.017** |
| | (0.010) | (0.020) | (0.006) | (0.008) |
| County-by-Season FE | x | x | x | x |
| Summer-by-Year FE | x | x | x | x |
| State-by-Year FE | x | x | | |
| County-by-Year FE | | | x | x |
| Detailed Weather Controls | | x | x | x |
| Only Counties With Ozone Monitors | | | | x |
| Weighted by Population | x | x | x | x |

**Notes:** Medication costs are reported in 2015 dollars and deflated using the US CPI for urban consumers. The entries in Table 3 are the coefficient estimates from the DDD estimator described in equation (7) when the dependent variable is the log of medication costs per person-season-year in a county. Each coefficient is from a separate regression that includes a full set of county*year, season*year, and county*season fixed effects. Additional control variables are listed in the bottom of Table 3. The reported standard errors are clustered at the state-season level. The regressions are GLS weighted by the square root of MarketScan population in a given county-year-season. The reported standard errors are clustered at the state-season level Total population refers to the 2,539 counties in the main sample. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***). Number of observations is as follows: Row 1 columns (1) to (3): 30,926. Row 1 column (4): 2,338. Row 2 columns (1) to (3): 28,784. Row 2 column (4): 2,324. Row 3 columns (1) to (3): 24,080. Row 3 column (4): 2,296.

**Table 4. Effect of the NBP Emissions Market on Mortality Rates**

| | (1) | (2) | (3) | (4) | (5) |
|---|---|---|---|---|---|
| **A: All Deaths** | | | | | |
| 1. All Deaths | -2.15** | -2.09 | -1.47* | -5.34*** | -2.24 |
| | (0.94) | (3.54) | (0.81) | (1.82) | (1.46) |
| **B: Specific Causes of Death** | | | | | |
| 2. Respiratory or Cardiovascular | -0.75 | -1.20 | -0.47 | -2.27* | -0.84 |
| | (0.49) | (1.79) | (0.67) | (1.17) | (1.03) |
| 3. Non-Respiratory and Non-Cardio. | -1.40** | -0.89 | -1.00** | -3.08*** | -1.40* |
| | (0.57) | (1.92) | (0.50) | (0.84) | (0.82) |
| 4. External | 0.57** | -0.02 | 0.32 | 0.28 | 0.37 |
| | (0.23) | (0.50) | (0.33) | (0.60) | (0.44) |
| **C. All Causes of Death, by Age Group** | | | | | |
| 5. Age 0 (Infants) | -2.26 | -8.85 | -4.24 | 3.63 | -11.07 |
| | (3.90) | (7.30) | (6.14) | (9.99) | (9.73) |
| Estimated Change in 2005 Deaths | -39 | -152 | -73 | 63 | -191 |
| 6. Ages 1-64 | -0.09 | -1.46 | -0.13 | -0.37 | -0.57 |
| | (0.32) | (1.07) | (0.47) | (1.14) | (0.84) |
| Estimated Change in 2005 Deaths | -105 | -1,701 | -151 | -431 | -664 |
| 7. Ages 65-74 | -1.85 | -15.41 | -2.25 | -9.58 | -1.23 |
| | (4.96) | (11.79) | (6.16) | (10.43) | (6.11) |
| Estimated Change in 2005 Deaths | -165 | -1,373 | -200 | -853 | -110 |
| 8. Ages 75+ | -39.31*** | -40.32* | -18.63* | -90.91*** | -23.40 |
| | (8.38) | (20.51) | (10.68) | (23.00) | (17.93) |
| Estimated Change in 2005 Deaths | -3,367 | -3,572 | -1,813 | -7,912 | -2,507 |
| County-by-Season FE | x | x | x | x | x |
| Summer-by-Year FE | x | x | x | x | x |
| State-by-Year FE | x | x | | | |
| County-by-Year FE | | | x | x | x |
| Detailed Weather Controls | | x | x | x | x |
| Counties With Ozone Monitors | | | | x | |
| Data Begins in 2001 | | | | | x |

**Notes:** The entries in Table 4 are the coefficient estimates from the DDD estimator described in equation (7) where the dependent variable is deaths per 100,000 population in each county-year-season. Each coefficient is from a separate regression that includes a full set of county*year, season*year, and county*season fixed effects. Additional control variables are listed in the bottom of Table 4. The reported standard errors are clustered at the state-season level. The regressions are GLS weighted by the square root of the relevant population in a given county-year-season. Unless noted otherwise, the data begins in 1997. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***). Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***). Number of observations is 55,858 for columns (1) to (3); 3,124 for column (4); and 35,546 for column (5).

**Table 5. Effect of $NO_x$ Emissions and Ambient Ozone Concentrations On Medication Purchases and Mortality: Ordinary Least Squares and Instrumental Variables Estimates, 2001-2007**

| | Log Medication Costs | | | All-Cause Mortality | | |
|---|---|---|---|---|---|---|
| | All Counties | Counties with $NO_x$ Emissions | Ozone Monitored Counties | All Counties | Counties with $NO_x$ Emissions | Ozone Monitored Counties |
| | (1) | (2a) | (2b) | (3) | (4a) | (4b) |
| **A: OLS** | | | | | | |
| NOx Emissions | 0.07 | 0.40 | --- | -0.71* | -0.76* | --- |
| | (1.13) | (1.09) | --- | (0.39) | (0.39) | --- |
| 8-Hour Ozone | --- | --- | 0.72 | --- | --- | 0.12 |
| | --- | --- | (1.00) | --- | --- | (0.22) |
| Days ≥65 ppb | --- | --- | 0.22 | --- | --- | 0.01 |
| | --- | --- | (0.19) | --- | --- | (0.05) |
| **B: 2SLS** | | | | | | |
| NOx Emissions | 21.20 | 12.01** | --- | 5.16 | 5.35* | --- |
| | (13.77) | (5.78) | --- | (3.85) | (2.99) | --- |
| 8-Hour Ozone | --- | --- | 6.23** | --- | --- | 2.62** |
| | --- | --- | (2.33) | --- | --- | (1.28) |
| Days ≥65 ppb | --- | --- | 1.69** | --- | --- | 0.86* |
| | --- | --- | (0.72) | --- | --- | (0.49) |

**Notes:** The coefficient estimates in columns (1), (2a), and (2b) are multiplied by 1000 for readability. All estimates are based on the 2001-2007 sample. $NO_x$ emissions are measured in thousand tons per county. All regressions include county*year, season*year, and county*season fixed effects, as well as the detailed weather controls. The regressions are GLS weighted by the square root of the relevant population in a given county-year-season (MarketScan or full population). In Panel B, the endogenous variable is NOx or ozone and the excluded instrument is Summer*Post*NBP interaction (see equation 9). Number of observations is 30,926 for medication regressions including all counties, 7,616 for medication regressions including counties with $NO_x$ emissions, 2,338 for medication regressions including only counties with ozone monitors, 35,546 for mortality regressions including all counties, 7,840 for mortality regressions including counties with $NO_x$ emissions, and 2,352 for mortality regressions only including counties with ozone monitors. The sample is smaller for medications than for mortality due to counties without no medication data or zero expenditures. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***).

## Table 6. The Welfare Impacts of the NBP and the Social Benefits of $NO_x$ and Ozone Reductions

| | Medication Costs ($ Million) (1) | Medication Copayments ($ Million) (2) | Mortality: Number of Deaths (3) | Mortality: Monetized Value ($ Million) (4) | Total Using (1) ($ Million) (5) | Total Using (2) ($ Million) (6) |
|---|---|---|---|---|---|---|
| **A. An Upper Bound Estimate of NBP's Social Costs** | | | | | | |
| Upper Bound Per Year | --- | --- | --- | --- | $1,076 | $1,076 |
| Upper Bound, 2003-2007 Total | --- | --- | --- | --- | $4,843 | $4,843 |
| **B. Estimates of the NBP's Benefits** | | | | | | |
| Total Per Year | $820 | $150 | 2,238 | $1,068 | $1,888 | $1,219 |
| Total 2003-2007 | $3,690 | $676 | 10,069 | $4,807 | $8,497 | $5,483 |
| **C. The Annual Social Benefits of $NO_x$ Reductions in NBP states (Million Tons)** | | | | | | |
| Regressions Based on . . . | | | | | | |
| All Counties | $1,086 | $211 | 7,791 | $3,720 | $4,806 | $3,931 |
| Counties with $NO_x$ Emissions | $615 | $121 | 8,117 | $3,875 | $4,491 | $3,996 |
| **D. The Social Benefits of Ozone Reductions in NBP States (ppb)** | | | | | | |
| 1 ppb Ozone Decrease | $319 | $47 | 3,326 | $1,588 | $1,907 | $1,635 |
| 1 Less Day With Ozone > 65 ppb | $87 | $13 | 1,072 | $512 | $599 | $525 |

**Notes:** All dollar amounts are in 2015 constant dollars deflated using BLS CPI for urban consumers. The mortality impact estimates without dollar signs are number of deaths. The monetized mortality impact uses the VSL of $2.27 million (2015 dollars) from Ashenfelter and Greenstone (2004) and the age adjustments from Murphy and Topel (2006, p. 888). The implied VSLs are as follows: $2.26 million (infants); $1.78 million (age 1-64); $0.7 million (age 65-74); $0.3 million (age 75+). Total 2003-7 decrease due to NBP assumes impact is for half of 2003 summer and for all of summers 2004-2007. NBP cost upper bound is based on the permit price of about $2,523/ton ($2015) and estimated total abatement quantity of 427,000 tons. The numbers in Panel A comes from multiplying together the mean NBP allowance price per ton, the effect of the NBP on county-level $NO_x$ emissions (Table 2, column 4), and the number of counties in the NBP states (1,185). The numbers in Panels B-D come from multiplying together regression estimates of how the NBP, $NO_x$, or ozone affects medication costs or mortality by the total number of people in the NBP states, averaged over 2003-2007 (136 million people). Specifically, Panel B, column (1), uses the estimate from Table 3, column (3). Panel B, columns (2)-(3) use the estimates from Table 4, column (3), Panel C. Panel C, column (1) uses the estimate from Table 5, column (1), Panel B. Panel C, columns (2)-(3) use the estimate from Table 5, Panel B, column (3). Panel D, column (1) uses the estimate from Table 5, Panel B, column (2). Panel D, columns (2)-(3) use the estimate from Table 5, Panel B, column (4). Panel D estimates are based on regressions using counties with ozone monitors. All estimates apply to the full population in NBP states. See the text for further details.

**ONLINE APPENDIX FOR:**

# Defensive Investments and the Demand for Air Quality: Evidence from the NOx Budget Program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## Appendix I: The $NO_x$ Budget Trading Program and Particulate Matter

This appendix provides one explanation based in atmospheric chemistry as to why the $NO_x$ Budget Trading Program might have limited effects on particulate matter. We begin by defining the relevant compounds:

$PM_{10}$ and $PM_{2.5}$: particulate matter
$NO_x$: nitrogen oxides
NO: nitric oxide, a component of $NO_x$
$NO_2$: nitrogen dioxide, a component of $NO_x$
$NH_4NO_3$: ammonium nitrate, the component of $PM_{2.5}$ and $PM_{10}$ which $NO_x$ can form
$NO_3$: nitrate, a derivative of $NO_x$
$NH_4$: ammonium
$SO_4$: sulfate, formed as a byproduct of electricity generation
$NH_{4e}$: excess ammonium, i.e., ammonium which remains after $NH_4$ has bonded with $SO_4$
$NH_3$: ammonia
$HNO_3$: nitric acid, a derivative of $NO_x$

A summary is that excess ammonium ($NH_{4e}$) is the necessary ingredient for nitrate ($NO_3$) to become ammonium nitrate ($NH_4NO_3$), which is a component of particulates. In the absence of $NH_{4e}$, $NO_x$ and $NO_3$ do not form particulate matter. $NH_{4e}$ levels were low in the Eastern U.S. during the operation of the $NO_x$ Budget Trading Program because levels of sulfate ($SO_4$) were high enough to absorb much of the available $NH_4$, so that little sulfate remained to bond with nitrate.

A more detailed explanation follows. For $NO_x$ to become a component of $PM_{10}$ or $PM_{2.5}$, $NO_x$ must decompose to nitrate ($NO_3$). Nitrate then must undergo a reaction with excess ammonium ($NH_{4e}$) to form ammonium nitrate ($NH_4NO_3$). Ammonium nitrate is a component of particulate matter but nitrate is not. So a necessary condition for $NO_x$ to increase particulate matter is the presence of sufficient excess ammonium to convert nitrate into ammonium nitrate.

To assess the empirical relevance of this explanation, we calibrated an air quality model (CRDM) using the 2002 National Emissions Inventory, as in Muller and Mendelsohn (2012). According to calculations from CRDM, the Eastern U.S. had relatively low levels of $NH_{4e}$ during the operation of the $NO_x$ Budget Trading Program. Excess ammonium levels were low in part because $NH_4$ preferentially bonds with $SO_4$, which is a byproduct of sulfur emissions. Even with the Acid Rain program, sulfur levels were high enough in the Eastern U.S. in 2003-2007 that little $NH_4$ remained as $NH_{4e}$ after the $NH_4$-$SO_4$ reaction occurred.

According to calculations using CRDM, in the period 2003-2007, the Eastern U.S. had relatively low levels of excess ammonium, which could explain why we fail to find consistent evidence consistently that the $NO_x$ Budget Program affected particulate levels. Pandis and Seinfeld (2006), a widely-cited atmospheric chemistry text, note that this phenomenon is well-established:

> "The formation of ammonium nitrate is often limited by the availability of one of the reactants. Figure 10.24 shows the ammonium concentration as a function of the total

available ammonia and the total available nitric acid for a polluted area. The upper left part of the figure (area A) is characterized by relatively high total nitric acid concentrations and relatively low ammonia. Large urban areas are often in this regime. The isopleths are almost parallel to the y-axis in this area, so decreases in nitric acid availability do not affect significantly the $NH_4NO_3$ concentration in this area." (p. 483)

## Appendix II: Medications as Defensive Investments

It is worth considering the extent to which medication purchases studied in this paper should be defined as defensive, rather than just another health expenditure. We argue that whether the drugs are taken before or after health conditions manifest themselves, the important issue is that drugs aim to mitigate a negative health condition in both instances. Indeed, a recent *New England Journal of Medicine* article underscores that the key to treating asthma is limiting exposure to ambient concentrations of air pollution *and* consumption of pharmaceuticals:

> Achieving good long-term control of asthma (infrequent asthmatic symptoms, an unrestricted level of activity, normal or near-normal lung function, and rare asthmatic attacks requiring emergency care) requires a multifaceted approach: avoidance of environmental stimuli that can provoke bronchoconstriction and acute and chronic airway inflammation . . . *and* [emphasis added] drug therapy (Fanta 2009, p. 1005).

Further there has been a substantial decline in asthma mortality and emergency department visits since the 1970s and 1980s, even as the incidence of asthma increased; a leading explanation is the more widespread use of asthma medications, specifically inhaled corticosteroids (Fanta 2009). The point is that these drugs reduce the incidence of negative health outcomes and that the failure to account for expenditures on them in the previous literature has led to downward biased measures of willingness to pay for ozone reductions.

Additionally, we believe the theoretical economics literature supports the interpretation of the medication results. Specifically in the canonical model of health production function (see e.g., the recent *JEL* review by Graff-Zivin and Neidell 2013), individual compensatory responses to ambient pollution levels are typically decomposed into averting activities (aimed at reducing the amount of ingested pollution) and mitigating activities (aimed at reducing the negative effect of ingested pollution), with the latter including healthcare costs and medication purchases. In this framework, it is clear from the model structure that medication purchases are a defensive measure, not an outcome.

While these arguments suggest that all medications are defensive, we do provide one estimate which distinguishes respiratory medications (which are taken after respiratory symptoms appear) from long-term control medications, which are taken regularly in order to prevent the appearance of symptoms. Using National Drug Codes, we attempted to distinguish "maintenance" respiratory medications that are taken every day or week to treat chronic respiratory conditions, from "rescue" respiratory medications that are taken once acute respiratory symptoms appear. We distinguish these medications using lists from Fanta (2009) of which respiratory medications are short-acting (i.e., rescue) versus long-acting (i.e., control). We also list a set of results which restricts regressions to mail-order purchases, since these are more likely to be focused on long-term control medications. We find evidence that the NBP decreased purchases of both short-acting and long-term control medications (Appendix Table 2). The point estimates for the overall sample indicate that the NBP decreased purchase of short-acting medications by 2.4 percentage points and decreased purchase of long-term control medications by 2.1 percentage points; only the long-term control estimate is statistically significant. The estimated coefficients are slightly larger though less precise in the subset of counties with ozone monitors. Our estimates for mail-order purchases are similar to the overall estimates of NBP decreasing expenditure by about 1.5 percentage points.

## Appendix III: Data and Methodology Details

This appendix provides additional information on the study sample construction and the definitions of the main variables.

### Population Denominator

The main medication variable is the log of medication purchases per person. Because not all persons purchase medications, we count the number of individuals eligible to purchase medications from the enrollment files. We only count individuals who have a variable indicating that they recorded drug purchases, though this covers essentially all individuals in the firms and years we study. When the enrollment file has a missing value for an individual's county, we fill it with the county which is reported for that available in the nearest available lag; or if no lagged months have a reported county, then we fill it with the closest-available lead. We use the first month of a season to count the population in that season.

### Medication Purchases

The MarketScan data report the purchase county, date, the medication's National Drug Code (NDC), and the money paid from consumer and insurer to the medication provider. 2007 is the last year of the MarketScan dataset available for this analysis, so that is the last year of data for the analysis. Reported medication payments vary substantially across individuals and include some negative values. To address measurement error in medication payments, we measure the mean medication payment for each National Drug Code. This calculation uses all positive recorded medication payments for observations in the main regression sample. The dependent variable in main regressions is the log of this mean price. We deflate all currency to real year 2015 values using the BLS CPI for urban consumers. Medication purchases are assigned to the county where they are recorded. We include only medication purchases from individuals recorded in the enrollment files as working for one of the firms in the sample. While the main analysis includes individuals in a balanced panel of 19 firms, sensitivity analyses in Appendix Tables 2 and 4 report estimates from a balanced panel of about 600,000 persons in these firms. For confidentiality reasons MarketScan does not identify the 19 firms. These tables also report a sensitivity analysis using the 11 firms which appear in all the years 2000-2007.

It is worth noting that the medication expenditure estimates reported in Table 6 potentially suffer from re-transformation bias since the regression models are log-linear in expenditures, but Table 6 reports untransformed expenditures. When we apply the "smearing" estimator of Duan (1983) that corrects for re-transformation bias we obtain essentially the same estimates as the unadjusted ones in Table 6.

As discussed in the main text, the medication estimates are representative of Americans employed in large firms and their dependents, who appear in the MarketScan data; these people may have better baseline health than the average American, but may also have better health insurance and hence spend more on medications than the average American. Appendix Table 6 compares the characteristics of individuals in the MarketScan database with individuals in the January 2003 Current Population Survey (CPS) and the year 2000 Census. Compared to CPS respondents, MarketScan workers are about 11 percentage points more likely to be in a union, 7 percentage points more likely to receive a salary (rather than working for hourly wages), and 5 percentage points more likely to be full-time (rather than part-time). Heavy industry sectors are heavily over-represented in MarketScan, relative to workers in the CPS, while the retail, finance, insurance, real estate, and services are substantially under-represented in MarketScan. The age distribution of MarketScan individuals is similar to the overall age distribution in the year 2000 census, except MarketScan excludes individuals aged 65 and older.[1]

### Hospitalizations

As mentioned in the main text, we classify hospitalizations according to the mode procedure for each visit. MarketScan codes the primary diagnosis for each service provided in a hospitalization episode. Patients admitted to the hospital typically receive numerous services, some of which may have distinct diagnosis codes.

### Weather

We compiled weather data from records of the National Climate Data Center Summary of the Day files (File TD-3200). The key control variables for our analysis are the daily maximum and minimum temperature, total daily precipitation, and dew point temperature. In order to ensure accurate weather readings, and complete county-day data files, we construct our weather variables for a given year from the readings of all weather stations that report valid readings for every day in that year. The acceptable station-level data is then aggregated at the county level by taking an inverse-distance weighted average of all the valid measurements from stations that are located within a 200 km radius of each county's centroid, where the weights are the inverse of their squared distance to the centroid so that more distant stations are given less weight. This results in complete weather by county-day files that we can link with the other files in our analysis.

Dew point temperature values are limited in the raw data. To address missing values, we impute them by regressing observed county-day dew point temperature on mean temperature, mean precipitation, year fixed effects, county fixed effects, a quartic polynomial in day-of-year, and interactions of the quartic polynomial separately with each of the following three variables: daily mean precipitation, daily mean temperature and precipitation*temperature. The regression uses 1997-2007 data. We replace missing dew point temperature values with these imputed values.

### Pollution

We assign each ton of emitted pollution to the county where the emitting source is located. Counties with no recorded emissions are assigned emissions of zero. We use this approach because we observe all NBP-regulated pollution emissions and the information on their emitting source, so we record them as such without any kind of spatial averaging needed.

[1] Given the large sample sizes and differences in means, hypothesis tests reject equality of these characteristics between MarketScan and the other data sources at 99% significance level.

We convert pollution units using values from Spellman and Whiting (2005). For each pollutant, we calculate ambient levels in each monitor-day, then the unweighted average across monitors in each county-day, and finally aggregate up to county-season. All ambient pollution regressions are GLS based on the square root of the total number of underlying pollution readings.

The abrupt beginning and end of the market on May 1 and October 1 makes a daily regression discontinuity estimator seem appealing. However, because ozone in the Eastern US mainly reaches high levels in July and August, the market is likely to have small effects on ambient pollution on April 30 or October 1. Although emitted pollution changed sharply around these dates (Figure 1), we detect no change in mean daily ambient pollution in small windows around these dates.

We did explore statistical models that separately estimate effects of the market on pollution (and health) outcomes in each month of summer. These specifications did not have statistical power to distinguish effects in different months of summer, and hence we focus on results that treat summer as homogenous. Modeling the market's impact on summer overall, rather than month-by-month, also produces medium-term estimates of the market's impact. This makes the results less susceptible to the concern that changes in air quality cause short-term displacement of mortality or medication purchases without changing their medium- or long-run values.

**Econometric Approach**

The instrumental variables estimates including all counties are computationally demanding given the three sets of fixed effects (county*year, county*season, season*year) and approximately 2,500 counties in the data. We implement these estimators using efficient optimization routines (Guimaraes and Portugal 2010). In smaller samples, these routines obtain numerically equivalent point estimates to those of conventional methods. The estimated confidence regions may be slightly conservative, since in these smaller samples they obtain standards errors that are a few percent larger than those estimated with conventional methods.

The fact that our counterfactual allows other changes such as $NO_x$ regulations to be operating in the background can be seen from Appendix Figure 2. In both seasons and regions of the country, $NO_x$ emissions were declining even before the NBP began, and for 2000-2002 the pre-trends were similar in both regions and seasons. The counterfactual analyzed here is if summertime $NO_x$ emissions in the Eastern U.S. continued along the pre-trends observed before 2002, but did not experience the large 2002-2004 decline which the NBP generated.

## Appendix IV: Additional Details about the $NO_x$ Budget Trading Program

This Appendix describes additional details about the $NO_x$ Budget Trading Program (NBP) not explained in the main text.

Our research design is based on comparing emissions in summer versus winter months. Because $NO_x$ abatement technologies have substantial operating costs (Fowlie 2010), units begin operating them around May 1 and stop around September 30. Part of the operating cost comes from the "heat rate penalty" of selective catalytic reduction—the fact that they require a small amount of electricity to operate.

The NBP grew out of the Ozone Transport Commission (OTC), an organization of Northeast States formed in the 1990s. OTC studies found that ozone levels the Northeast U.S. had high ozone partly because prevailing winds transported $NO_x$ from the industrial Midwest to the Northeast, where it produced ozone in the Northeast (OTC 1998). The OTC led to a version of the NBP that operated in 1999-2002 and produced small declines in summer $NO_x$ emissions.[2] The OTC then created a more stringent version of the NBP which began in 2003 and operated until 2008.[3]

As described in the main text, the NBP included 19 states plus DC. Georgia was initially slated to enter the market in 2007 but the EPA eventually chose to exclude Georgia.

Policymakers included some provisions to help smooth the start of the market. Regulators provided an additional set of initial allowances in 2003-2004 known as the Compliance Supplement Pool or CSP to help states begin compliance with the market without threatening electricity supply reliability. Ultimately many of these allowances were banked to future years. Unused allowances from the NBP could be transferred to the CAIR ozone season program which succeeded the NBP after 2008.

In 2002, summertime emissions from sources participating in this market totaled approximately 1 million tons, with a significant downward pre-trend that had similar magnitude in both the East and West (Appendix Figure 2). Compared to the level of $NO_x$ emissions in 2002, the final cap of 550,000 tons would have decreased emissions by 45%. As discussion of our results later in the paper shows, however, accounting for the pre-trend and the fact that emitters banked allowances

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[2] This market also goes under the name $NO_x$ SIP Call. This smaller market also operated in May-September. The OTC market aimed to decrease summer NOx emissions by 76,000 tons (OTC 2003). NOx emissions from regulated NBP units in our data fell by 504,000 tons between 2002 and 2005, or about 6.6 times more than the OTC market. While in principle this earlier market could be a source of confounding variation for the pollution and mortality regressions which begin in 1997, those regressions have similar signs and significance as the pollution and mortality regressions beginning in 2001. The OTC market cap for most states did not change between 2000 and 2002, so this is not a potential source of confounding variation for our pre-period in those years. The only small change in the OTC market in these years is that some pollution sources in Maryland and DC entered the market in those years, and the cap in those states modestly increased to accommodate them (OTC 2003).

[3] 2007 is the last year of the MarketScan dataset available for this analysis, so that is the last year of data for the analysis. In 2009, the Clean Air Interstate Rule (CAIR) replaced this market. In 2010, the EPA proposed a Transport Rule which would combine this $NO_x$ market with a market for $SO_2$ emissions. In July 2011, the EPA replaced this proposal with the Cross-State Air Pollution Rule, which regulates power plant emissions in 27 states with the goal of decreasing ambient ozone and particulate levels.

across years shows that the causal impact of the market was to decrease emissions by only 35-39 percent.

One important question involves the geographic scope of the NBP's effects, and of our analysis. As discussed in the main text, the main analysis excludes states adjacent to the NBP region from the main results because their treatment status is ambiguous, though sensitivity analyses consider alternatives. The main analysis excludes Wisconsin, Iowa, Missouri, Georgia, Mississippi, Maine, New Hampshire, and Vermont. We do not exclude Arkansas or Florida because they share only small sections of border with the NBP area and because prevailing winds blow to the Northeast, away from these states. We exclude Maine even though it does not share a border with the NBP region because it is downwind and close to many NBP states. We define Alabama as an NBP state even though the southern region of the state did not participate in the market.

These exclusions have a basis in prevailing wind patterns and directions. On the 96 percent of days in the NBP region where windspeeds are below 6 meters per second, ozone and its precursors travel less than 300 miles (Husar and Renard 1997). This implies that on many days, emissions from the NBP region affect the states we exclude, but do not affect the states we include in the comparison group. Husar and Renard (1997) find that ozone and precursors travel up to 120 miles on days with windspeeds below 3 m/s and up to 300 miles on days with windspeeds below 6 m/s. We obtained raw windspeed readings from the National Climate Data Center's Summary of the Day – First Order (DSI-3210) files and measured average windspeed and directions across states for all states in the NBP region. Mean windspeeds are below 3 m/s on 61 percent of days and are 3-6 m/s on 34 percent of days. Although prevailing winds blow to the East, on many days wind blows in other directions. On 27 percent of days wind primarily blows to the north, on 35 percent of days it primarily blows to the East, on 21 percent to the South, and on 17 percent to the West.

Another interesting question is the extent to which any changes in $NO_x$ or other pollution emissions occurred due to fuel conversion of units, for example from coal to natural gas. This is ambiguous from participation data available from the EPA's Air Markets Program Data, though the data do rule out large-scale closure of coal units. The number of coal units in the NBP actually grew from 845 in 2003 to 856 in 2008. At the same time, the number of gas units grew from 1,168 to 1,305 and the number of oil units fell from 471 to 463. So the absolute number of coal units rose, while the proportion of NBP units that are coal fell by 1.5 percentage points. Qualitatively similar patterns occur within the subset of NBP units that are owned by electric utilities, and within the subset that are industrial boilers. These statistics ignore the 1% of units that report multiple primary fuels.

We also explored whether the $NO_x$ reductions produced any counterproductive outcomes, with mixed results. When an area has low concentrations of volatile organic compounds relative to $NO_x$, then decreasing $NO_x$ can <u>increase</u> ozone levels. First, we identify a list of such "VOC-constrained" cities from Blanchard (2001). Second, we define a county as VOC-constrained if its mean ratio of weekend/weekday ozone exceeds 1.05. The former approach finds that the change in ozone concentrations is similar in VOC-constrained and -unconstrained regions. The latter indicates that in VOC-constrained regions of the NBP, the decline in ozone was smaller than in the unconstrained areas. See rows 10 and 11 of Appendix Table 1.

While Figure 2 (A) shows the regression-adjusted event study graph of $NO_x$ emissions, Appendix Figure 2 shows the raw emissions trends separately by season and year. Appendix Figure 2 (A) shows that the NBP led to sharp and discontinuous reductions in summer emissions in the Eastern U.S., starting in 2003 when the market began in 8 Northeastern states and Washington, DC. Emissions declined another 15-20 percent starting in May 2004, when the market added 11 more Eastern states. Winter emissions in the Eastern U.S. continued their gradual downward pre-2003 trend. In contrast, Appendix Figure 2 (B) reveals that summer and winter $NO_x$ emissions in the non-NBP states evolved smoothly over time, with similar downward trends and no evidence of any discernible trend change in 2003 and 2004, when the NBP was implemented. In short, this Appendix Figure shows that $NO_x$ emissions declined in exactly the areas, months, and years that the market design would predict.

## Appendix V: Additional Sensitivity Analyses

All of the ambient and emitted pollution results are further evaluated and probed in Appendix Table 1, which considers a wide range of specifications, including changes in the method used to compute the standard errors and alternative sample selection rules. In addition, we estimated models that also allowed for differential pre-existing trends in the NBP states during the summer. In general, the models fail to reject the null of no difference in pre-existing trends and cause the standard error on the parameter of interest, $\gamma_1$, to increase by a factor of 2 to 3. The only substantive change is that the impact on ozone concentrations is larger in magnitude although the 95% confidence intervals of the estimates from specifications with and without the differential trends overlap.

Appendix Table 2 reports medication results from a series of robustness checks, none of which alter the qualitative conclusions from Table 3. For example, Row 12 shows that defining the dependent variable as the log of copayments per person rather than as the log of mean medication costs per person results in a decrease in medication costs per person of 1.4 percent, as opposed to the 1.5 percent effect in the main sample. Estimates for the subsample of young children are very imprecise due in part to the smaller sample of children.

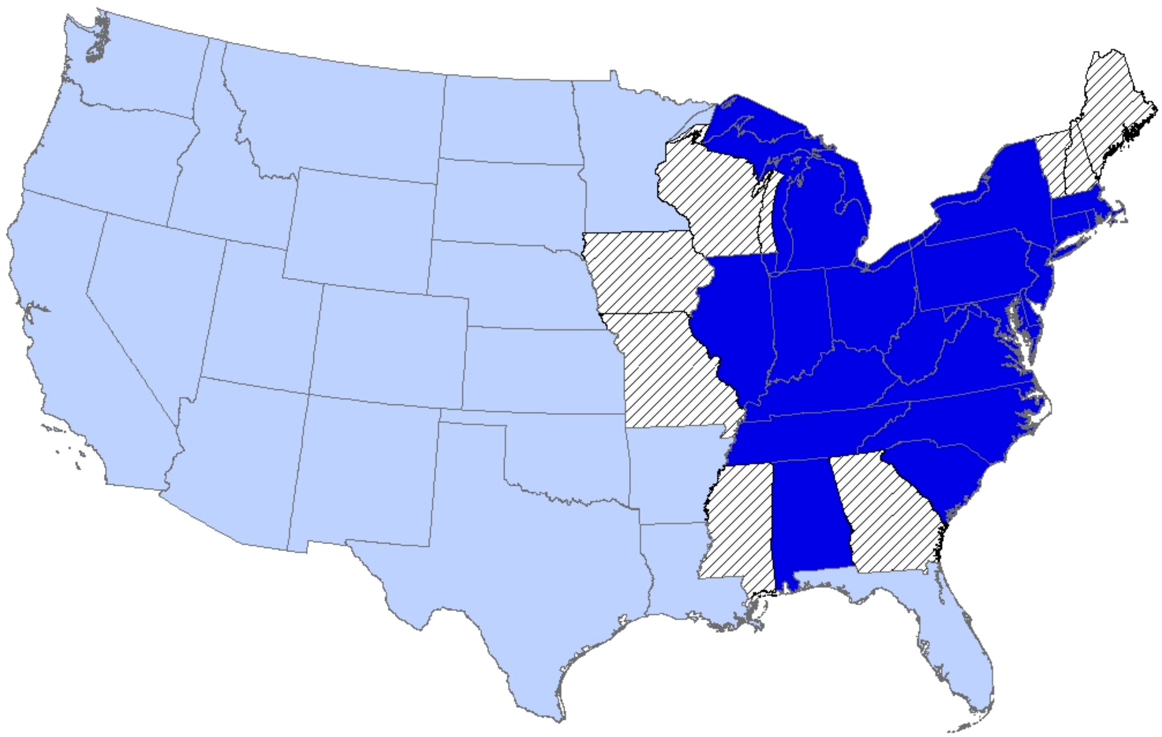
Row 2 considers the non-NBP states excluded from the main analysis sample since they border NBP states and are expected to benefit to due wind transmission of pollution. This group includes Wisconsin, Iowa, Missouri, Georgia, Mississippi, Maine, New Hampshire, and Vermont. Our results are similar when we reassign these states to the NBP.

Appendix Table 4 discusses similar sensitivity analyses for the effect of the NBP on mortality. The qualitative conclusions are similar under these alternative estimates.

Appendix Table 5, column (1) explores heterogeneity in the magnitude of the NBP's effect on ambient ozone across different sub-parts of the NBP. Ten northeastern states entered the NBP market in 2003 whereas other states began in 2004 (the delay was due to litigation). Also because prevailing winds blow to the Northeast, they might experience larger effects of emissions decreases in the industrial Midwest. The NBP decreased ozone in these states by an additional 2 ppb, decreased medication expenditures in these states by an additional 3.8 percentage points. The NBP caused a larger decline in ozone for the ten northeastern states which entered the market in 2003, which is unsurprising since these states entered the market a year earlier than other states did and since prevailing winds blow to the northeast. The NBP also caused a larger decline in ozone for counties which had relatively high ozone in 2002, which fits with the finding of Figure 2 (B) that most of the decline was from days with the highest ozone levels.

## Appendix VI: Supplementary Figures and Tables

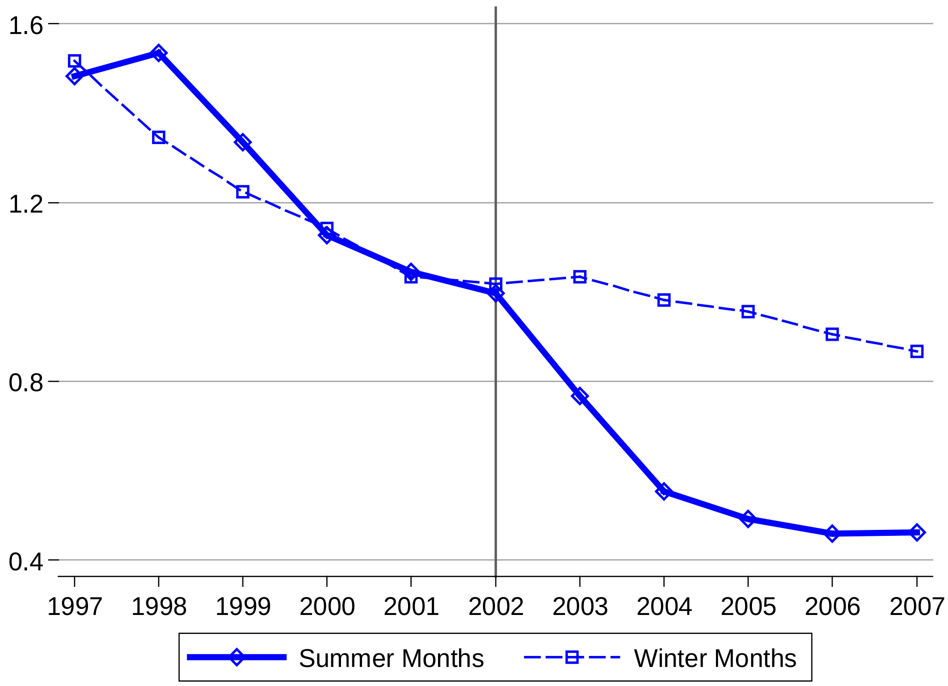
**Appendix Figure 1. Participation in NBP by State**



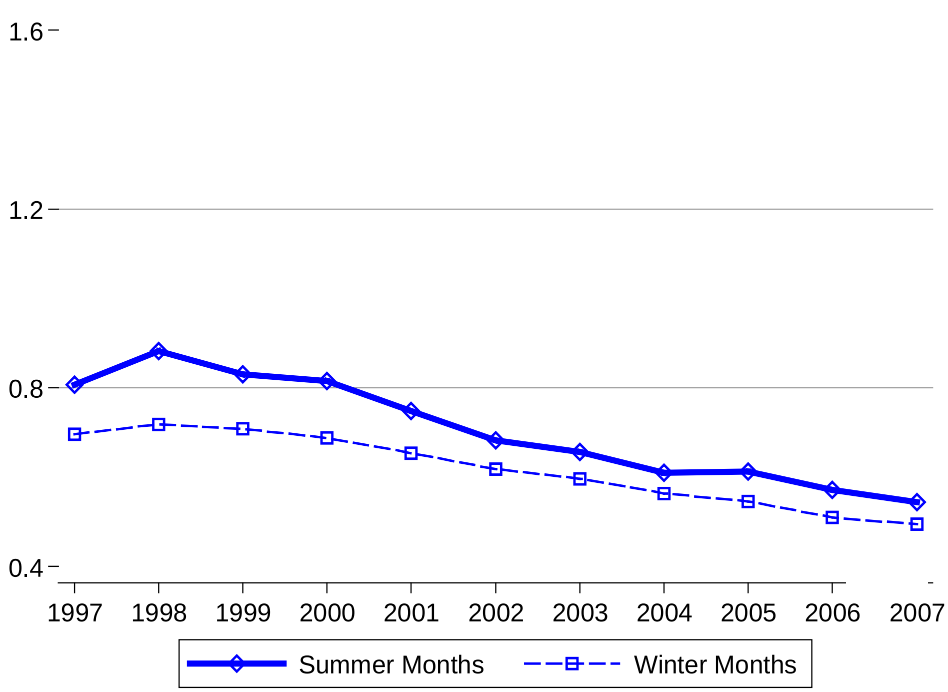
**Notes:** Dark blue states are participating in NBP during the 2003-2007 period (referred as 'NBP states' in the text). Light blue states are not participating (non-NBP states). Shaded states are excluded from the main analysis sample.

**Appendix Figure 2. Summer-Equivalent Seasonal $NO_x$ Emissions (Mil. Tons)**

**(A) States Participating in NBP**



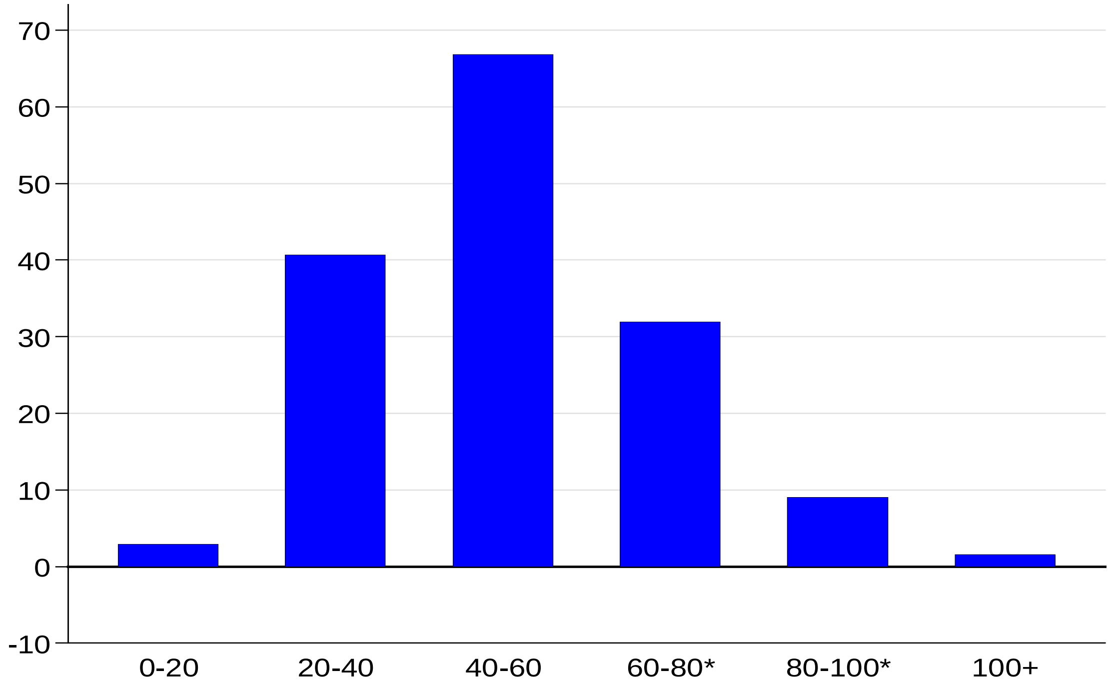

**(B) States Not Participating in NBP**



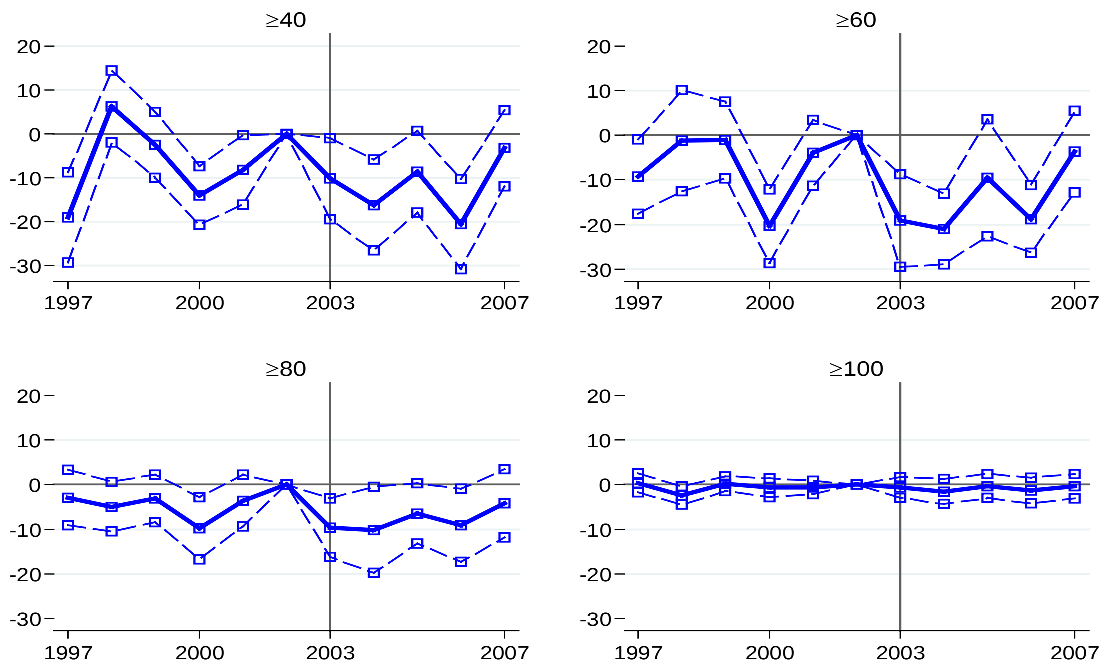

**Notes:** The data underlying Appendix Figure 2 is expressed as summer-equivalent since the summer period has 5 months while the winter period has 7 months. Specifically, the summer equivalent of winter emissions is actual winter emissions multiplied by 5/7. These graphs show summary statistics describing total emissions, not regression results. Summer defined as May-September, winter as January-April and October-December. NBP participating states include: Alabama, Connecticut, Delaware, District of Columbia, Illinois, Indiana, Kentucky, Maryland, Massachusetts, Michigan, Missouri, New Jersey, New York, North Carolina, Ohio, Pennsylvania, Rhode Island, South Carolina, Tennessee, Virginia, and West Virginia. States not participating in NBP include: Arkansas, Arizona, California, Colorado, Florida, Idaho, Kansas, Lousiana, Minnesota, Montana, Nebraska, Nevada, New Mexico, North Dakota, Oklahoma, Oregon, South Dakota, South Carolina, Texas, Utah, Washington, Wyoming. Alaska, Georgia, Hawaii, Iowa, Maine, Mississippi, Missouri, New Hampshire, Vermont, and Wisconsin are excluded from the main analysis sample.

**Appendix Figure 3. NBP Market Impact on Ambient Ozone Pollution, Detail**

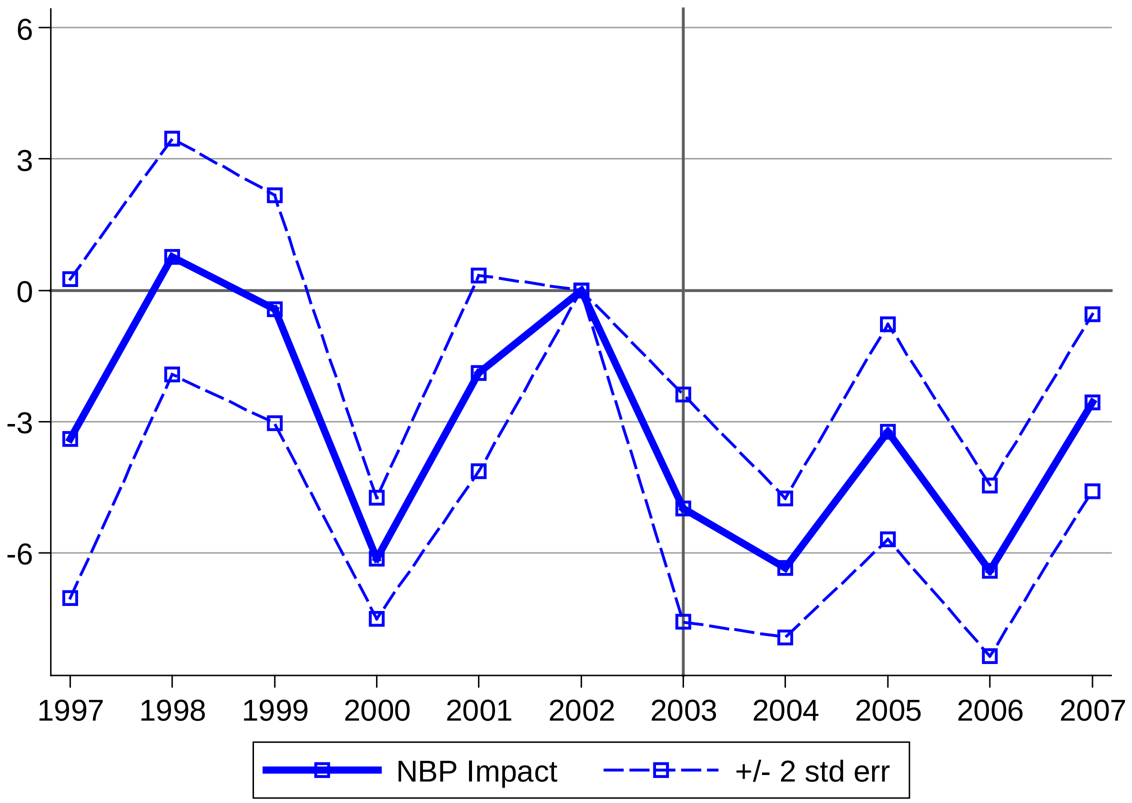
**(A) Number of Summer Days in 6 Ozone Bins, NBP Participating States, 2001-2002**




**(B) NBP Market Impact on Number of Summer Days with Ozone above 40, 60, 80 or 100 ppb**



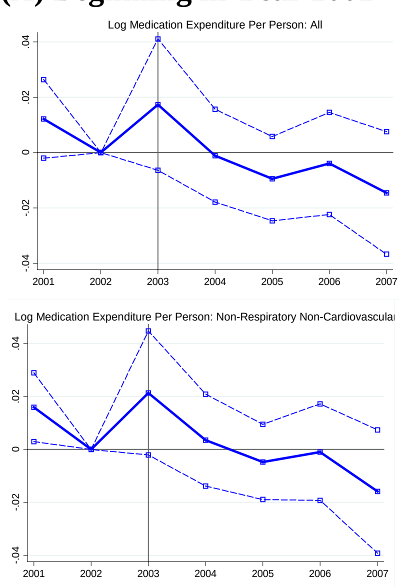
**(C) Event Study for Daily Ozone 8-Hour Values, 1997-2007**



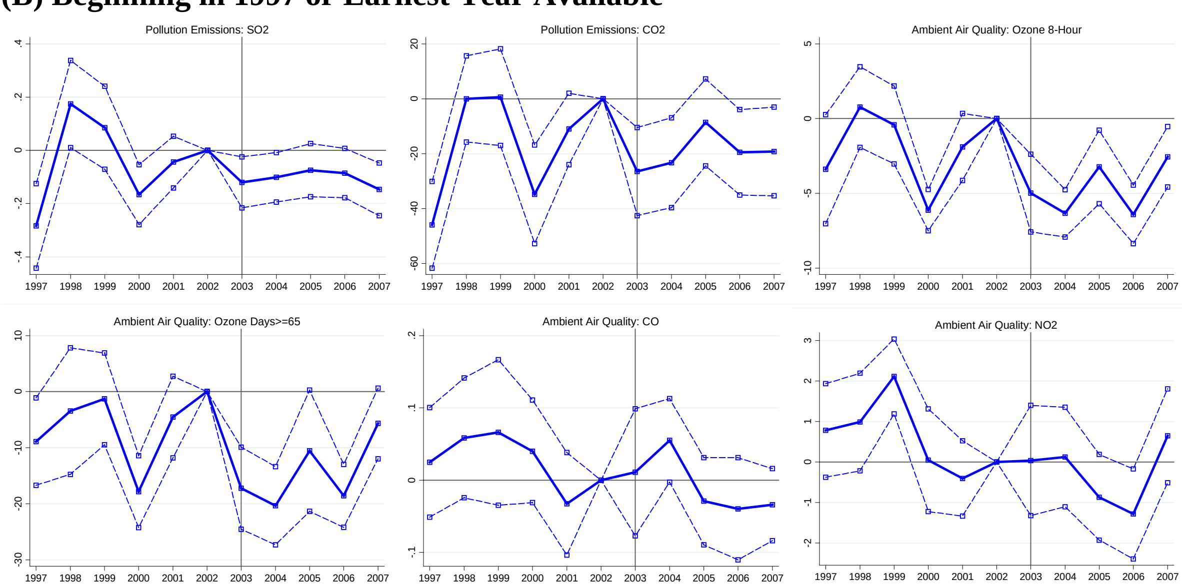

**Notes:** Ozone 8-hour value is measured as the maximum rolling 8-hour mean of hourly values within in each day, which is the statistic used in EPA nonattainment designations. Panel A shows the average number of summer days (out of a possible 153 days) in 6 bins for daily ozone 8-hour value in the NBP states in 2001-2002 (pre-NBP period). Panel B shows the estimated impact of NBP on the number of summer days in 4 of these categories for daily ozone 8-hour value. Panel C shows the coefficients from an event study regression for ozone 8-hour values where the estimates for year 2002 are restricted to have a value of 0. All regressions include detailed weather controls and a full set of county*year, season*year, and county*season fixed effects, and are weighted by the number of ozone monitors in each county. The standard errors underlying the 95% confidence intervals (dashed lines) are clustered at the state-season level.

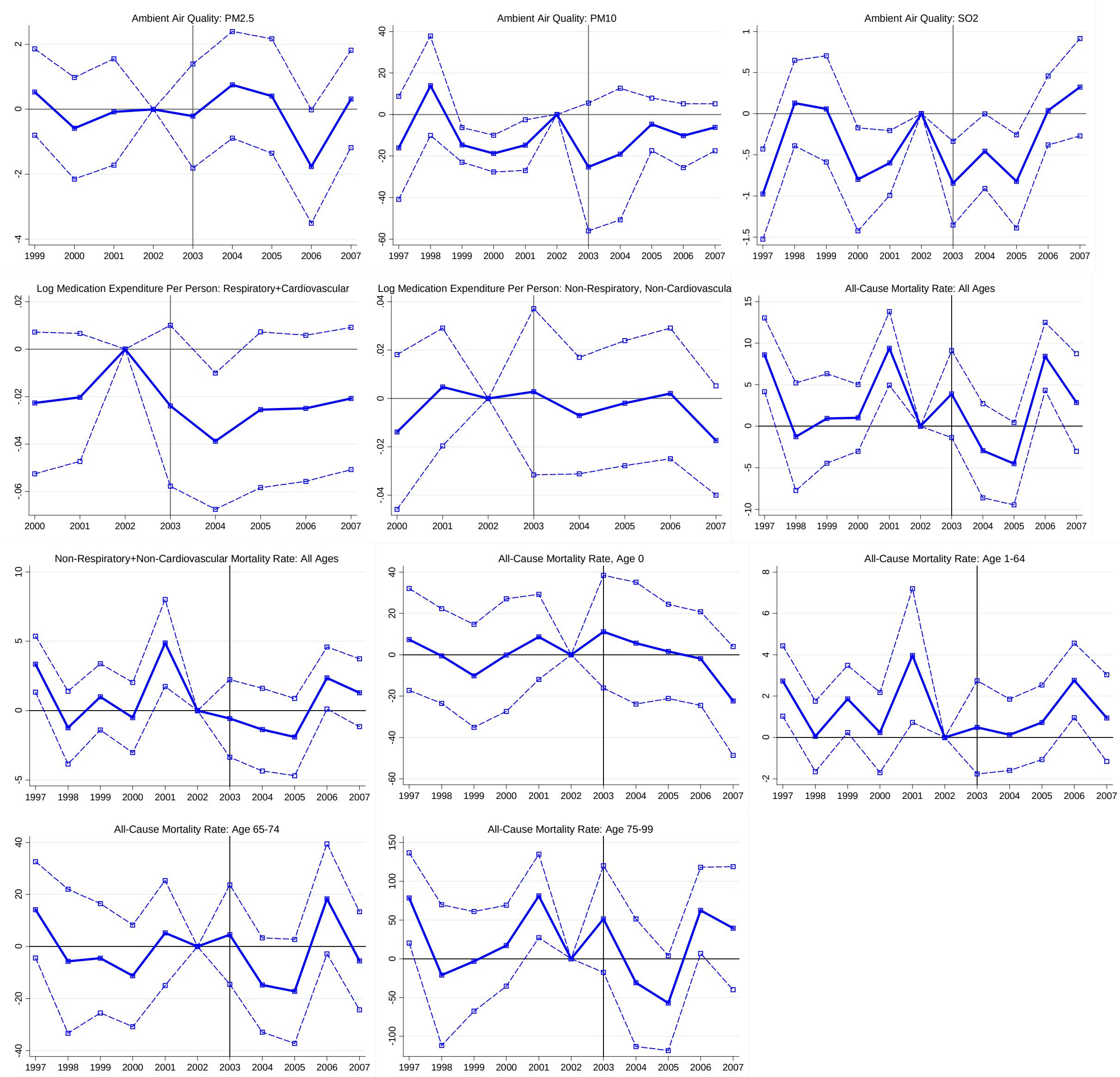
# Appendix Figure 4. Event Study Graphs for All Outcomes

## (A) Beginning in Year 2001




## (B) Beginning in 1997 or Earliest Year Available






**Notes:** Appendix Figure 4 reports the coefficients from event study regressions for all outcomes where the estimates for year 2002 are restricted to have a value of 0. All regressions include detailed weather controls and a full set of county*year, season*year, and county*season fixed effects, and are weighted by the relavant variable for a specific outcome (number of ozone monitors in each county, population in each county). The standard errors underlying the 95% confidence intervals (dashed lines) are clustered at the state-season level. See Appendix Figure 1 notes or text for NBP participation status designation. The standard errors underlying the 95% confidence intervals (dashed lines) are clustered at the state-season level.

## Appendix Table 1. Sensitivity Analysis: Emitted and Ambient Pollution

| | Emitted Pollution | | | Air Quality (Ambient Pollution) | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | $NO_x$ | $SO_2$ | $CO_2$ | Ozone | Ozone Days ≥65ppm | CO | $PM_{2.5}$ | $PM_{10}$ | $SO_2$ | $NO_2$ |
| | (1) | (2) | (3) | (4) | (5) | (6) | (7) | (8) | (9) | (10) |
| 1. Baseline Sample | -0.33*** | -0.07** | -12.43* | -3.06*** | -8.95*** | -0.02 | -0.31 | -1.18 | 0.13 | -0.92** |
| State-Season Clusters | (0.07) | (0.03) | (6.59) | (0.50) | (2.61) | (0.02) | (0.31) | (0.87) | (0.18) | (0.35) |
| County Clusters | (0.08) | (0.05) | (7.76) | (0.52) | (2.44) | (0.03) | (0.31) | (1.16) | (0.24) | (0.46) |
| State Clusters | (0.10) | (0.04) | (9.38) | (0.70) | (3.68) | (0.03) | (0.43) | (1.24) | (0.25) | (0.49) |
| State-Year Clusters | (0.05) | (0.04) | (6.63) | (1.13) | (3.62) | (0.03) | (0.48) | (1.44) | (0.18) | (0.38) |
| County-Season Clusters | (0.05) | (0.04) | (5.49) | (0.37) | (1.74) | (0.02) | (0.22) | (0.83) | (0.17) | (0.33) |
| Firm-County Clusters | (0.07) | (0.05) | (7.96) | (0.53) | (2.33) | (0.03) | (0.29) | (1.17) | (0.23) | (0.42) |
| 2. Counties With Ozone Monitors | -0.22** | -0.25 | -61.55 | -3.06*** | -8.95*** | -0.01 | -0.29 | -3.43 | 0.16 | -0.98* |
| | (0.10) | (0.21) | (43.28) | (0.50) | (2.61) | (0.03) | (0.38) | (4.56) | (0.24) | (0.55) |
| 3. Non-NBP Border States Assigned to NBP | -0.25*** | -0.05 | -10.34* | -3.03*** | -8.90*** | -0.02 | -0.34 | -0.40 | 0.11 | -0.77** |
| | (0.06) | (0.03) | (5.83) | (0.49) | (2.56) | (0.02) | (0.29) | (0.93) | (0.18) | (0.36) |
| 4. Limit to Most Comparable Non-NBP States | -0.32*** | -0.04 | -6.55 | -2.97*** | -9.91*** | -0.03 | -0.45 | -1.73 | 0.14 | -1.08*** |
| | (0.06) | (0.03) | (7.27) | (0.53) | (2.89) | (0.02) | (0.31) | (1.23) | (0.24) | (0.34) |
| 5. Post = 1.0 in Year 2003 | -0.32*** | -0.09*** | -16.00** | -3.65*** | -10.70*** | -0.02 | -0.52 | -1.30 | 0.01 | -0.96** |
| | (0.06) | (0.03) | (6.67) | (0.65) | (2.66) | (0.02) | (0.37) | (0.93) | (0.16) | (0.36) |
| 6. Post = 0.0 in Year 2003 | -0.27*** | -0.04 | -6.56 | -1.98*** | -5.77*** | -0.01 | -0.09 | -0.84 | 0.20 | -0.71** |
| | (0.06) | (0.03) | (5.35) | (0.40) | (2.14) | (0.02) | (0.24) | (0.77) | (0.16) | (0.29) |
| 7. Drop Year 2003 | -0.35*** | -0.08** | -14.46** | -3.86*** | -12.02*** | -0.02 | -0.51 | -0.69 | 0.05 | -1.05*** |
| | (0.07) | (0.03) | (6.79) | (0.61) | (2.83) | (0.02) | (0.35) | (0.90) | (0.19) | (0.37) |
| 8. Diff-in-Diff for NBP States | -0.32*** | -0.07* | -4.26 | -4.72*** | -11.04*** | 0.04** | 0.43* | -1.88** | 0.04 | -1.23** |
| | (0.06) | (0.04) | (3.87) | (0.43) | (1.99) | (0.02) | (0.24) | (0.87) | (0.12) | (0.50) |
| 9. Monitors Operating ≥ 30 Weeks | --- | --- | --- | -3.04*** | -11.72*** | -0.02 | -0.46 | -0.43 | 0.11 | -0.62 |
| | --- | --- | --- | (0.42) | (1.89) | (0.02) | (0.29) | (0.97) | (0.17) | (0.37) |
| 10. Main Effect | --- | --- | --- | -2.85** | -12.21** | --- | --- | --- | --- | --- |
| | --- | --- | --- | (1.22) | (4.27) | --- | --- | --- | --- | --- |
| Summer*Post*NBP *VOC-Constrained | --- | --- | --- | 0.19 | 1.44 | --- | --- | --- | --- | --- |
| | --- | --- | --- | (1.25) | (4.80) | --- | --- | --- | --- | --- |
| 11. Main Effect | --- | --- | --- | -3.52*** | -11.60*** | --- | --- | --- | --- | --- |
| | --- | --- | --- | (0.57) | (2.83) | --- | --- | --- | --- | --- |
| Summer*Post*NBP* (High Weekend $O_3$) | --- | --- | --- | 1.29** | 7.21*** | --- | --- | --- | --- | --- |
| | --- | --- | --- | (0.51) | (2.29) | --- | --- | --- | --- | --- |

**Notes:** The entries in Appendix Table 1 are the coefficient estimates on the Summer*Post*NBP variable from separate DDD regressions (unless noted otherwise). The reported standard errors are clustered at the state-season level (unless noted otherwise). The regressions use the specification and sample of Table 2 column (4) (unless otherwise noted). The entries after row 1 present different levels of clustering for standard errors. Row 3 takes eight non-NBP states that border the NBP area (Iowa, Georgia, Maine, Missouri, Mississippi, New Hampshire, Vermont, and Wisconsin) and assigns them to the NBP area. Row 4 limits non-NBP states to the half with ozone data which have the smallest Euclidean distance from NBP states, defined from year 2002 mean ozone, $NO_x$ emissions per square mile, medication costs per capita, and temperature. The non-NBP comparison states selected by this criterion are: Arkansas, California, Colorado, Kansas, Nevada, New Mexico, Oklahoma, and Texas. "Monitors Operating ≥ 30 weeks" uses a monitor selection rule which requires each monitor to have valid readings in 30 weeks of each year in the data, rather than the 47-week rule used in the main results. "Summer*Post*NBP*VOC-Constrained" reports the interaction of the main triple-difference term with an MSA indicator for being VOC constrained based on Blanchard (2001). "Summer*Post*NBP*(High Weekend O3)" interacts the main triple-difference term with an indicator for whether the weekend/weekday ozone ratio of a county exceeds

1.05. This provides an alternative indicator of VOC-constrained regions. Regressions use 2001-2007 data. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***).

## Appendix Table 2. Sensitivity Analysis: Medications

| | All | Respiratory or Cardiovascular | Non-Respiratory and Non-Cardiovascular |
|---|---|---|---|
| | (1) | (2) | (3) |
| 1. Baseline Sample | -0.015** | -0.021*** | -0.014** |
| State-Season Clusters | (0.006) | (0.008) | (0.006) |
| County Clusters | (0.007) | (0.007) | (0.007) |
| State Clusters | (0.009) | (0.011) | (0.009) |
| State-Year Clusters | (0.006) | (0.008) | (0.007) |
| County-Season Clusters | (0.005) | (0.005) | (0.005) |
| Firm-County Clusters | (0.006) | (0.007) | (0.007) |
| 2. Non-NBP Border States Assigned to NBP | -0.016*** | -0.021*** | -0.015*** |
| | (0.005) | (0.007) | (0.005) |
| 3. Limit to Most Comparable Non-NBP States | -0.014* | -0.016* | -0.013 |
| | (0.008) | (0.009) | (0.008) |
| 4. Post = 1.0 in Year 2003 | -0.010 | -0.015* | -0.008 |
| | (0.006) | (0.008) | (0.007) |
| 5. Post = 0.0 in Year 2003 | -0.016*** | -0.021*** | -0.016*** |
| | (0.005) | (0.006) | (0.005) |
| 6. Drop Year 2003 | -0.012* | -0.019** | -0.011* |
| | (0.006) | (0.008) | (0.006) |
| 7. Log Medications (Not Costs) | -0.014** | -0.020** | -0.012** |
| | (0.006) | (0.008) | (0.006) |
| 8. Ages 0-17 | 0.005 | 0.001 | 0.004 |
| | (0.013) | (0.020) | (0.013) |
| 9. Panel of People | -0.009 | -0.018** | -0.005 |
| | (0.008) | (0.009) | (0.008) |
| 10. Levels (Not Logs) | -10.640*** | -2.991*** | -8.149*** |
| | (2.445) | (0.950) | (2.187) |
| 11. Purchase-Specific Costs | -0.013** | -0.020*** | -0.011* |
| | (0.006) | (0.007) | (0.006) |
| 12. Copay | -0.014** | -0.022*** | -0.012* |
| | (0.006) | (0.008) | (0.006) |

**Appendix Table 2. Sensitivity Analysis: Medications (ctd)**

| | All | Respiratory or Cardiovascular | Non-Respiratory and Non-Cardiovascular |
|---|---|---|---|
| | (1) | (2) | (3) |
| 13. 2000-2007 Firm Panel | -0.009 | -0.017** | -0.008 |
| | (0.006) | (0.008) | (0.007) |
| 14. Mail Order | -0.015** | -0.009 | -0.004 |
| | (0.006) | (0.006) | (0.007) |
| 15. Only Counties with Ozone Monitors | -0.016*** | -0.017* | -0.017** |
| | (0.006) | (0.010) | (0.008) |
| 16. Respiratory: Short-Acting Only | --- | -0.025 | --- |
| | --- | (0.018) | --- |
| 17. Respiratory: Long-Term Only | --- | -0.020** | --- |
| | --- | (0.008) | --- |

**Notes:** The entries in Appendix Table 2 are the coefficient estimates on the Summer*Post*NBP variable from separate DDD regressions using data for 2001-2007. The reported standard errors are clustered at the state-season level. Row 2 takes eight non-NBP states that border the NBP area (Iowa, Georgia, Maine, Missouri, Mississippi, New Hampshire, Vermont, and Wisconsin) and assigns them to the NBP area. Row 3 limits non-NBP states to the half with ozone data which have the smallest Euclidean distance from NBP states, defined from year 2002 mean ozone, $NO_x$ emissions per square mile, medication costs per capita, and temperature. The non-NBP comparison states selected by this criterion are: Arkansas, California, Colorado, Kansas, Nevada, New Mexico, Oklahoma, and Texas. “Log Medications (not costs)” uses counts of medication purchases, rather than cost measures. "Panel of People" uses the much smaller panel of persons who appear in all observations of the MarketScan sample. "Levels (Not Logs)" specifies the response variable in levels rather than logs. "Purchase-Specific Costs" uses the raw reported prices, rather than averaging across national drug codes to deal with outliers as in the main analysis. "Counties with Ozone Data" restricts the analysis to include only counties with ozone monitors satisfying the monitor selection rule. "Copay" measures costs as purchase-level patient expenditures. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***).

**Appendix Table 3. Sensitivity Analysis: Hospitalization Costs**

| | (1) | (2) | (3) | (4) |
|---|---|---|---|---|
| **A: All Hospitalizations** | | | | |
| 1. All Hospitalizations | -5.91 | -4.17 | -7.43 | -79.12*** |
| | (19.60) | (22.34) | (23.12) | (28.01) |
| **B: Specific Groups of Hospitalizations** | | | | |
| 2. Respiratory or Cardiovascular | -9.26 | -9.13 | -10.22 | -44.95*** |
| | (5.59) | (6.77) | (6.69) | (13.30) |
| 3. Non-Respiratory and Non-Cardiovascular | 3.36 | 4.96 | 2.79 | -34.17 |
| | (15.98) | (17.17) | (18.90) | (20.44) |
| County-by-Season FE | x | x | x | x |
| Summer-by-Year FE | x | x | x | x |
| State-by-Year FE | x | x | | |
| County-by-Year FE | | | x | x |
| Detailed Weather Controls | | x | x | x |
| Only Counties With Ozone Monitors | | | | x |
| Weighted by Population | x | x | x | x |

**Notes:** The entries in Appendix Table 3 are the coefficient estimates on the Summer*Post*NBP variable from separate DDD regressions using data for 2001-2007. The reported standard errors are clustered at the state-season level. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***).

**Appendix Table 4. Sensitivity Analysis: Mortality**

| | All | Respiratory or Cardiovascular | Non-Respiratory and Non-Cardiovascular |
|---|---|---|---|
| | (1) | (2) | (3) |
| 1. Baseline Sample | -1.47* | -0.47 | -1.00** |
| State-Season Clusters | (0.81) | (0.67) | (0.50) |
| County Clusters | (1.19) | (0.79) | (0.78) |
| State Clusters | (1.16) | (0.96) | (0.71) |
| State-Year Clusters | (1.70) | (1.13) | (0.88) |
| County-Season Clusters | (0.84) | (0.56) | (0.55) |
| 2. Non-NBP Border States Assigned to NBP | -0.73 | -0.25 | -0.48 |
| | (0.83) | (0.65) | (0.50) |
| 3. Limit to Most Comparable Non-NBP States | -0.34 | -0.06 | -0.28 |
| | (0.83) | (0.72) | (0.47) |
| 4. Post = 1.0 in Year 2003 | -0.93 | 0.13 | -1.06** |
| | (0.79) | (0.65) | (0.49) |
| 5. Post = 0.0 in Year 2003 | -1.78** | -1.00 | -0.78* |
| | (0.79) | (0.63) | (0.46) |
| 6. Drop Year 2003 | -1.64* | -0.61 | -1.02* |
| | (0.84) | (0.67) | (0.53) |
| 7. Logs (Not Levels) | -0.01** | -0.01** | 0.00 |
| | (0.00) | (0.00) | (0.00) |
| 8. Age-Adjustment | -1.45 | -0.71 | -0.74 |
| | (0.89) | (0.69) | (0.54) |
| 9. Only Counties With Ozone Monitors | -5.34*** | -2.27* | -3.08*** |
| | (1.82) | (1.17) | (0.84) |

**Notes:** The entries in Appendix Table 4 are the coefficient estimates on the Summer*Post*NBP variable from separate DDD regressions using data for 1997-2007. The reported standard errors are clustered at the state-season level. Row 2 takes eight non-NBP states that border the NBP area (Iowa, Georgia, Maine, Missouri, Mississippi, New Hampshire, Vermont, and Wisconsin) and assigns them to the NBP area. Row 3 limits non-NBP states to the half with ozone data which have the smallest Euclidean distance from NBP states, defined from year 2002 mean ozone, $NO_x$ emissions per square mile, medication costs per capita, and temperature. The non-NBP comparison states selected by this criterion are: Arkansas, California, Colorado, Kansas, Nevada, New Mexico, Oklahoma, and Texas. "Logs (Not Levels)" specifies the response variable in logs rather than levels. Age-adjustment modifies the response variable to use age-adjusted mortality counts, rather than total deaths per population. Regressions use 1997-2007 data. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***).

**Appendix Table 5. Heterogeneity Within NBP Region in NBP Effects**

| | 8-Hour Ozone<br>(1) | All Medication Expenditures<br>(2) | All Mortality<br>(3) |
|---|---|---|---|
| Interaction of NBP*Post*Summer with Dummy for … | | | |
| 1. Ten Northeastern States | -2.648*** | -0.038*** | -1.118 |
| | (0.794) | (0.014) | (2.273) |
| 2. County has Below-Median Weekend/Weekday Ozone Ratio in Summer 2002 (VOC-Constrained) | -0.364 | 0.015 | 4.793 |
| | (0.904) | (0.009) | (4.373) |
| 3. County has Above-Median Post-2003 Mean Summer Temperature | 2.630** | -0.005 | -0.655 |
| | (1.135) | (0.006) | (1.633) |
| 4. State has Above-Median Child Asthma Rate | -1.104 | 0.029*** | 2.765 |
| | (1.067) | (0.010) | (3.334) |
| 5. State has Above-Median Adult+Child Asthma Rate | -1.109 | -0.013 | -0.474 |
| | (1.167) | (0.012) | (3.274) |
| 6. County's has Above-Median 2002 Medication Expenditure or Mortality Per Capita | -0.975 | -0.006 | -1.601 |
| | (0.792) | (0.009) | (4.445) |
| 7. County has Above-Median 2002 Respir.+Cardio. Medication Expenditure or Mortality per Capita | --- | -0.043 | -1.120 |
| | --- | (0.043) | (3.361) |
| 8. County has Above-Median Number of Ozone Days ≥60ppb in Summer 2002 | -3.098*** | -0.004 | -7.047* |
| | (0.488) | (0.012) | (4.078) |
| County-by-Season FE | x | x | x |
| Summer-by-Year FE | x | x | x |
| County-by-Year FE | x | x | x |
| Detailed Weather Controls | x | x | x |
| Data Begin in 2001 | x | x | x |
| Weighted by Population | x | x | x |

**Notes:** The entries in Appendix Table 4 are the coefficient estimates on Summer*Post*NBP*X, where X is the interaction term specified in each row of the table. The reported standard errors are clustered at the state-season level. The regression also controls for Summer*Post*NBP (coefficient not shown), for detailed weather controls, and for county-by-season, county-by-year, and season-by-year fixed effects.
Row 1 interacts the main effect with an indicator for being in one of the ten Northeastern states where NBP began in 2003 rather than 2004. Row 3 interacts the main effect with a dummy for a summer having above-median post-2002 season-mean temperature, where the median is calculated separately for each

county. Rows 4 and 5 interact the main effect with dummies for a state having above-median 2002 asthma rates. Rows 6 and 7 interact the main effect with a dummy for a county having above-median summer 2002 medication expenditures or mortality. All data include years 2001-2007. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***).

**Appendix Table 6. Characteristics of the MarketScan Sample**

| | MarketScan (1) | CPS Jan 2003 (2) | Census 2000 (3) |
|---|---|---|---|
| Employee Classification | | | |
| Union | 0.24 | 0.13 | --- |
| Non-Union | 0.76 | 0.87 | --- |
| Salary | 0.60 | 0.53 | --- |
| Hourly | 0.40 | 0.47 | --- |
| Employment Status | | | |
| Active Full-Time Employee | 0.97 | 0.82 | 0.42 |
| Active Part-Time Employee | 0.03 | 0.18 | 0.11 |
| Relation of Patient to Employee | | | |
| Employee | 0.44 | --- | --- |
| Spouse | 0.23 | --- | --- |
| Child/Other | 0.33 | --- | --- |
| Industry | | | |
| Oil & Gas Extraction, Mining | 0.01 | 0.00 | 0.00 |
| Manufacturing, Durable Goods | 0.27 | 0.09 | 0.05 |
| Manufacturing, Nondurable Goods | 0.19 | 0.04 | 0.03 |
| Transportation, Communications, Utilities | 0.33 | 0.08 | 0.05 |
| Retail Trade | 0.02 | 0.12 | 0.07 |
| Finance, Insurance, Real Estate | 0.04 | 0.07 | 0.04 |
| Services | 0.14 | 0.44 | 0.28 |
| Sex | | | |
| Male | 0.48 | 0.52 | 0.49 |
| Female | 0.52 | 0.48 | 0.51 |
| Age | | | |
| 0-4 Years | 0.06 | 0.00 | 0.07 |
| 5-17 Years | 0.20 | 0.02 | 0.19 |
| 18-64 Years | 0.75 | 0.95 | 0.62 |
| ≥65 Years | 0.00 | 0.03 | 1.00 |

**Notes:** Column (1) describes the main 2001-2007 sample. Current Population Survey (CPS) data restricted to individuals with strictly positive working hours.

**Appendix Table 7. Effect of $NO_x$ Emissions and Ambient Ozone Concentrations On Medication Purchases and Mortality: Instrumental Variables Estimates, 2001-2007, by Cause**

| | Log Medication Costs | | | All-Cause Mortality | | |
|---|---|---|---|---|---|---|
| | All Counties | Counties with $NO_x$ Emissions | Ozone Monitored Counties | All Counties | Counties with $NO_x$ Emissions | Ozone Monitored Counties |
| | (1) | (2a) | (2b) | (3) | (4a) | (4b) |
| **A: 2SLS, Respiratory or Cardiovascular** | | | | | | |
| NOx Emissions | 29.41 | 16.33* | --- | 1.94 | 1.98 | --- |
| | (18.27) | (8.36) | --- | (2.36) | (1.48) | --- |
| 8-Hour Ozone | --- | --- | 6.68* | --- | --- | 1.18 |
| | --- | --- | (3.58) | --- | --- | (0.86) |
| Days ≥65 ppb | --- | --- | 1.81* | --- | --- | 0.39 |
| | --- | --- | (1.06) | --- | --- | (0.31) |
| **B: 2SLS, Non-Respiratory and Non-Cardiovascular** | | | | | | |
| NOx Emissions | 19.90 | 11.57** | --- | 3.22 | 3.37* | --- |
| | (13.43) | (5.60) | --- | (2.24) | (1.81) | --- |
| 8-Hour Ozone | --- | --- | 6.64** | --- | --- | 1.45*** |
| | --- | --- | (2.93) | --- | --- | (0.53) |
| Days ≥65 ppb | --- | --- | 1.80** | --- | --- | 0.48** |
| | --- | --- | (0.86) | --- | --- | (0.21) |

**Notes:** In Panel A, the dependent variable includes respiratory or cardiovascular medication costs or mortality; in Panel B, the dependent variable includes all non-respiratory and non-cardiovascular medication costs or mortality. The coefficient estimates in columns (1), (2a), and (2b) are multiplied by 1000 for readability. All estimates are based on the 2001-2007 sample. $NO_x$ emissions are measured in thousand tons per county. All regressions include county*year, season*year, and county*season fixed effects, as well as the detailed weather controls. The regressions are GLS weighted by the square root of the relevant population in a given county-year-season (MarketScan or full population). The endogenous variable is NOx or ozone and the excluded instrument is Summer*Post*NBP interaction (see equation 9). Number of observations is 30,926 for medication regressions including all counties, 7,616 for medication regressions including counties with $NO_x$ emissions, 2,338 for medication regressions including only counties with ozone monitors, 35,546 for mortality regressions including all counties, 7,840 for mortality regressions including counties with $NO_x$ emissions, and 2,352 for mortality regressions only including counties with ozone monitors. The sample is smaller for medications than for mortality due to counties without no medication data or zero expenditures. Asterisks denote p-value < 0.10 (*), <0.05 (**), <0.01 (***).

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