CLEARING THE AIR? THE EFFECTS OF GASOLINE CONTENT REGULATION ON AIR QUALITY

RUNNING TITLE: GASOLINE CONTENT REGULATION

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Abstract

This paper examines whether U.S. gasoline content regulations, which impose substantial costs on consumers, have successfully reduced ozone pollution. We take advantage of spatial and temporal variation in the regulations' implementation to show that federal gasoline standards, which allow refiners flexibility in choosing a compliance mechanism, did not improve air quality. This outcome occurred because minimizing the cost of compliance does not reduce emissions of those compounds most prone to forming ozone. In California, however, we find that precisely targeted, inflexible regulations requiring the removal of particularly harmful compounds significantly improved air quality.

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Since the passage of the original Clean Air Act in 1963, U.S. state and federal governments have implemented numerous policies designed to reduce human exposure to ground-level ozone pollution. Ozone is an odorless, colorless gas that has been linked to asthma, increased susceptibility to pneumonia and bronchitis, and damage to crops and natural

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vegetation. Michelle L. Bell *et al.* (2004) have estimated that even marginal short-term changes in ozone concentrations can have substantial human mortality impacts.[1] While ozone is not emitted directly by any source, the two classes of chemicals that react in the atmosphere to produce ozone—volatile organic compounds (VOCs) and oxides of nitrogen (NO_x)—are pollutants produced in part through human activity. Despite more than 40 years of emissions regulation, however, many areas of the U.S. continue to experience ambient air concentrations of ozone that exceed standards set by the Environmental Protection Agency (EPA).

This paper examines the effectiveness of one particular set of regulations: restrictions on the chemical composition of gasoline that are primarily intended to reduce VOC emissions from mobile sources. These regulations have recently come under scrutiny because they impact gasoline prices through two mechanisms: (1) an increase in production costs; and (2) segmentation of the U.S. gasoline market that leads to increased price volatility and increased opportunities for refiners to exert local market power. Gasoline content standards are not uniform across the country; the EPA regulates some states and counties more tightly than others, and some areas have implemented their own standards that are more stringent than those set by the EPA. The resulting "patchwork" of regulation prevents gasoline transporters from arbitraging price differences across areas with different gasoline standards. Several recent papers—Jennifer Brown *et al.* (2008), Ujjayant Chakravorty and Céline Nauges (2008), and Erich Muehlegger (2006)—have found that this market segmentation significantly increases both gasoline price levels (beyond the increase in production cost) and price volatility. The U.S. Congress, concerned about the price effects of segmented gasoline markets, inserted language into the

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[1] Bell *et al.* (2004) estimate that a 10 parts per billion (ppb) increase in ozone concentrations (relative to average concentrations of about 60 ppb) would result in 3,767 additional premature deaths annually across 95 urbanized areas in the U.S. For evidence regarding ozone's morbidity and environmental impacts, see EPA (2006), Enrico Moretti and Matthew Neidell (forthcoming), and Matthew Neidell (2004, 2009).

Energy Policy Act of 2005 (section 1541(b)) that constrained the ability of the EPA to enact or approve new gasoline standards that could exacerbate segmentation.

Despite the considerable attention paid to the price effects of gasoline content regulation, we are not aware of any study that comprehensively assesses the extent to which regulation has yielded benefits through reduced ozone pollution. While several different forms of regulation have been implemented—all primarily targeting VOC emissions—it is not known whether some types of regulation are more effective than others, nor whether some locations are more likely to benefit from regulation. Our objective in this study is therefore to address these issues using a panel dataset of ambient ozone concentrations across the United States.

Our primary finding is that the effectiveness of gasoline regulations varies substantially with the flexibility with which refiners are permitted to respond. A set of federal gasoline regulations that limits the total evaporation of VOCs from gasoline—without regard to which VOCs are most reactive in forming ozone—is estimated to have no economically or statistically significant effect on ground-level ozone concentrations. We attribute this result to the flexibility permitted by the regulations: refiners, free to choose which VOCs to remove from their gasoline, reduce the concentration of a type of VOC that is only weakly related to ozone formation. Refiners do not reduce concentrations of highly reactive VOCs because doing so is expensive and the overall VOC standard provides no compensating incentive or mandate.

In contrast, California’s gasoline regulations place strict content limits on precisely those VOCs that are most important in forming ozone, thereby eliminating refiners’ ability to avoid costly abatement of these compounds. As a result, California has enjoyed a significant improvement in air quality: we estimate that the introduction of California reformulated gasoline reduced ground-level ozone concentrations by 16% in the severely polluted Los Angeles – San

Diego area. A conservative back of the envelope calculation indicates that the benefits from this air quality improvement outweigh the regulation's cost based on mortality impacts alone.

These divergent outcomes speak to a tradeoff inherent in setting the degree of flexibility of environmental regulations. Flexible regulatory approaches, such as the federal VOC requirements for gasoline, are designed to reduce abatement costs relative to more restrictive command-and-control regulations by allowing firms to choose the least-cost compliance mechanism. However, flexibility may also result in reduced environmental benefits if different compliance mechanisms result in different patterns of emissions across space, time, or specific pollutants. This tradeoff seems unlikely to be unique to gasoline regulation. Tradable emission permit markets, for instance, have become popular because they allow compliance costs to be minimized across heterogeneous firms. Curtis Carlson *et al.* (2000), for example, find evidence of substantial abatement cost reductions stemming from trading of sulfur dioxide permits. However, standard permit trading systems may not achieve the first-best welfare outcome when the marginal benefits of abatement are spatially heterogeneous. Meredith Fowlie (2010), for instance, finds that the benefits of a major U.S. NO_x cap-and-trade program were undercut because heterogeneity in compliance incentives caused NO_x abatement to be concentrated in low marginal damage areas rather than dense urban centers in the Northeast. A more restrictive regulation that weighted emissions by local marginal damages—attenuating the ability of firms to flexibly adjust abatement across different regions—would have yielded a superior outcome.

In our setting, the impact of regulatory flexibility is particularly severe. While the flexible federal VOC standards result in lower compliance costs than does California's restrictive standard (1-1.5 cents per gallon vs. 8-11 cents per gallon),[2] they have no measurable effect on ozone pollution whereas California's regulation yields substantial declines in ozone concentrations. These results highlight the importance of anticipating firms' likely response when considering the implementation of flexible regulations, even in a command-and-control setting such as the design of gasoline content standards. To the extent that firms' behavioral responses may mitigate the desired environmental benefits of regulation, these welfare losses should be weighed against the compliance cost reductions conferred by a flexible regulatory design.

We identify the impacts of gasoline regulations on ozone by taking advantage of (1) daily measurements of ambient ozone concentrations from hundreds of air quality monitors across the United States during 1989-2003; (2) the rich spatial and temporal variation with which gasoline regulations were applied; and (3) the discrete nature with which these regulations were phased-in. Unlike, for example, standards for vehicle emissions control equipment that only produce effects gradually through turnover in the vehicle fleet, the adoption of a gasoline content standard immediately affects all vehicles on the road. We therefore seek to identify step changes in ozone concentrations at the times and locations in which gasoline regulations came into effect. We use two methods: (1) a difference-in-difference (DD) estimator that also controls for time-varying observables in treated vs. control areas; and (2) a regression discontinuity (RD) design that examines changes in ozone concentrations immediately before and after gasoline regulations came into effect. Our RD design is similar to that of Lucas Davis' (2008) study of Mexico City's

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[2] The 1-1.5 cents per gallon cost range for federal RVP is taken from Brown *et al.* (2008). The estimated 8-11 cents per gallon cost range for CARB comes from summing the 3 cents per gallon price effect of RFG from Brown *et al.* (2008) with a 5 to 8 cents per gallon estimated incremental cost of CARB (CARB, 2008).

driving restrictions in that we flexibly control for location-specific unobservables that may have affected changes in air quality over time.[3]

The remainder of the paper is organized as follows: section I provides a description of the gasoline reformulation policies examined in this paper. Section II describes the data we collected, and our identification strategy is given in section III. Section IV presents the estimation results. Section V discusses the role of refiners' behavior in explaining our findings, and section VI concludes.

I. Regulatory Background

A. Ground-level ozone formation

The primary goal of gasoline content regulation is to reduce ambient concentrations of ground-level ozone by targeting emissions of its chemical precursors: VOCs and NO_x. The chemical reactions through which VOCs and NO_x form ozone are complex in several ways that are important to gasoline regulation and this study. First, ozone formation requires warm temperatures and sunlight; gasoline content regulations therefore tend to be particularly stringent during the summer. Second, the ozone production function exhibits Leontief-like properties in its inputs of VOCs and NO_x. That is, in areas in which VOC concentrations are relatively high, ozone formation is "NO_x-limited" in that marginal reductions in VOC emissions will not affect ozone concentrations, but marginal reductions of NO_x emissions will. Conversely, areas with relatively high NO_x concentrations are said to be "VOC-limited." Third, and finally, VOCs include a large number of chemical compounds across which the reactivity in forming ozone varies considerably: some compounds are nearly 80 times more reactive than others. The

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[3] For other papers that measure the air quality and health benefits of U.S. air quality regulations, see Vernon J. Henderson (1996), Kenneth Y. Chay and Michael Greenstone (2003, 2005), and Michael Greenstone (2004).

effectiveness of gasoline content regulations may therefore hinge on whether they target VOCs or NO_x in a way that matches the VOC or NO_x limitations of specific geographic areas, and whether they are effective in reducing emissions of those VOCs that are particularly reactive in forming ozone.

VOCs and NO_x are also emitted by sources other than gasoline. Large NO_x emissions sources include electric generation plants, industrial boilers, and kilns. The use of solvents and paints contributes to VOC emissions, though the largest VOC sources are biogenic; deciduous trees are particularly significant contributors. It is because of these natural emissions that ozone formation in rural areas is generally NO_x-limited, while urban areas tend to be VOC-limited (Sanford Sillman 1999, Charles L. Blanchard 2001).[4] Finally, while NO_x emissions in one region can impact ozone levels up to 1000km downwind, VOC emissions only have a local effect (Jana B. Milford *et al.* 1989, Sillman 1999).

B. Reid vapor pressure (RVP) regulations

Gasoline regulations first targeted ground-level ozone pollution with the introduction of Reid vapor pressure (RVP) regulation in 1989. RVP, which is measured in pounds per square inch (psi), gauges the intensity with which VOCs are emitted from gasoline. These emissions may occur either on the road, via vehicles' exhaust, or through evaporation. For example, the fumes one smells when fueling a vehicle are evaporative VOC emissions. RVP regulation limits the RVP of gasoline sold during the summer months when hot, sunny weather is conducive to ozone formation. Refiners meet the RVP limits by reducing the concentration of "light" components—particularly butane—in the gasoline they sell (Tancred C.M. Lidderdale 1999).

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[4] There are exceptions. Rural areas immediately downwind of an urban plume are sometimes VOC-limited. Urban areas in the heavily-wooded South, such as Atlanta, can sometimes be NO_x-limited.

RVP regulation was introduced in two phases. Under phase I, which covered 1989 through 1991, each of the contiguous 48 states was assigned an RVP limit during the summer months. The limit varied by state according to EPA modeling of where VOC emission reductions were most needed, and took on a value of 10.5, 9.5, or 9.0 psi, with lower numbers indicating a tighter standard. The summer compliance period was June 1 – September 15 for retail gasoline stations and May 1 – September 15 for refiners and wholesale distribution terminals. RVP limits in some states were more stringent in July and August than in June and September, and Texas and Illinois had within-state differences in their limits. Table A1 in the appendix provides state-level detail of the RVP phase I program.[5]

RVP Phase II began in 1992 and required all counties in the United States to meet a summer RVP limit of at most 9.0 psi, which had been the most stringent limit under phase I. Moreover, RVP II mandated a tighter 7.8 psi summer RVP limit in southern states that were designated as being in nonattainment of the EPA's National Ambient Air Quality Standard for ozone. This tight limit reduces VOC emissions from gasoline by at least 15% relative to the 9.0 psi limit (EPA 1993). In addition, some areas implemented even stricter limits, as low as 7.0 psi, as part of their plans to meet the EPA's ozone standard. The entire state of California adopted an RVP standard of 7.8 psi. RVP phase II regulations remain active today, though in some areas they have been superseded by RFG or CARB standards, which are discussed below. RVP phase II details are provided in table A2 in the appendix. Figure 1 summarizes the timing with which RVP and subsequent regulations came into effect, and figure 2 provides a map of affected areas.

[Figure 1 – Approximately here]

[Figure 2 – Approximately here]

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[5] Details of the RVP phase I program, as well as RVP phase II and the RFG program, were extracted from the Code of Federal Regulations (40 CFR Part 80).

C. Reformulated gasoline (RFG) regulations

Federal reformulated gasoline (RFG) was mandated by the Clean Air Act Amendments of 1990, and the EPA began to enforce RFG regulations in 1995. RFG is federally mandated in areas designated to be in severe nonattainment of the EPA's ozone standard. Marginal, moderate, and serious nonattainment areas, which are not as polluted as severe areas but nonetheless fail to meet the EPA's ozone standard, may opt-in to federal RFG as part of their plans to reach attainment. A detailed listing of where and when RFG has been implemented is provided in table A3 in the appendix.

RFG was imposed in two phases: phase I came into force in 1995 and phase II in 2000. Like RVP regulations, RFG targets ground-level ozone; however, RFG regulations are tighter than those of RVP and involve both content criteria and performance standards. Under phase I, RFG must contain no more than 1% benzene, a toxic carcinogen that is also a VOC, and must contain at least 2% oxygen via use of an oxygenate such as MTBE or ethanol.[6] Phase I RFG must also reduce both VOC and toxic air pollutant (TAP) emissions by 15% relative to conventional gasoline. TAPs consist of five chemical compounds, including benzene, that are known carcinogens and are also VOCs.[7] In addition, the NO_x emissions of phase I RFG must not exceed those of conventional gasoline. The benzene, TAP, and NO_x standards are year-round, while the VOC standard applies only during the summer ozone season of June 1 – September 15. Phase II RFG tightened the seasonal VOC emission reduction standard to 25% while also

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[6] Oxygenates are used to reduce carbon monoxide (CO) emissions, particularly in the winter. In addition to oxygenate requirements through RFG, some non-RFG areas with CO pollution problems have their own oxygenate programs. The effectiveness of oxygenates in reducing ambient CO concentrations is not examined in this study.

[7] The TAPs are benzene, 1,3-butadiene, polycyclic organic matter, formaldehyde, and acetaldehyde. Of these, only benzene naturally occurs in gasoline; the others are combustion products. Benzene exhaust and non-exhaust (evaporative) emissions are estimated by the EPA to comprise 70-75% of all toxics emissions from gasoline. All five toxics are VOCs, though benzene is not strongly reactive in forming ozone.

tightening the TAP standard to 20%. In addition, phase II introduced a NO_x reduction requirement of 5.5% that applies year-round.

California and Arizona have implemented their own reformulated gasoline programs that are more stringent than federal RFG. Beginning in March 1996, California Air Resources Board (CARB) gasoline was required throughout the entire state of California. This requirement included rural parts of the state, most of which were in attainment of the EPA's ozone standard. Like federal RFG, CARB gasoline caps the benzene content of gasoline at 1% by volume. CARB gasoline targets VOC emissions more stringently than RFG, applying both a seasonal 7.0 psi RVP limit and year-round content criteria that limit concentrations of olefins (6% by volume) and aromatic hydrocarbons (25% by volume). Both of these classes of VOCs are highly reactive in forming ozone. CARB (2007) estimates that they are three to ten times more reactive than butane, the compound that refiners choose to remove from gasoline to meet federal RVP standards. In addition, CARB gasoline mandates an 80% reduction in sulfur content to reduce emissions of both sulfur dioxide and NO_x.[8] These standards are collectively more stringent than those of federal phase II RFG. Finally, Arizona's Cleaner Burning Gasoline (AZCBG) specifies that gasoline sold in the Phoenix area must meet federal RFG phase II specifications in the summer and CARB specifications in the winter.

II. Data

A. Air quality monitor data

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[8] We do not examine the impact of CARB's sulfur standards on sulfur dioxide concentrations in this study. The removal of sulfur affects NO_x emissions because sulfur inhibits vehicles' on-board NO_x emission control equipment.

We obtained data on ambient air concentrations of ozone from the EPA's Air Quality Standards database for 1989-2003.[9] This database reports hourly readings from the EPA's network of air quality monitors. We use these data to construct two measures of ozone concentrations at the monitor-day level: the daily maximum concentration and the daily 8 hour maximum. This latter measure is constructed by calculating the average ozone concentration within all 8 hour periods of each day, and then taking the maximum of these averages. We choose these two measures because the EPA's ozone standards have been built around them. The effective standard until June 2004 was a daily maximum ozone concentration of 0.12 parts per million (ppm); this was replaced by an 8 hour standard of 0.08 ppm. These standards reflect what the EPA believes are the maximum allowable ozone concentrations that protect public health.

We follow EPA data standards by disqualifying all monitor-days for which observations are not recorded for at least 9 hours between 9am and 9pm. We also disqualify monitor-years for which more than 25% of the days during the summer ozone season (1 June – 31 August) report no observation. Finally, we drop from the dataset monitors located in a county that is adjacent to a county treated with a more stringent regulation. Conversations with EPA staff have indicated that federal RFG is sometimes sold in non-RFG counties that border RFG areas to reduce fuel distribution costs; thus, such counties may not be true controls. Dropping these monitors reduces the dataset by 9.8% and does not substantially affect the estimated results.

Table 1 describes the sample of monitors used in this study. We possess measured ozone concentrations for a total of 1,144,025 monitor-days. The number of monitors increases over our sample frame from 720 to 945, indicating that the monitoring network grows at a rate of 2.0%

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[9] Data are available beyond 2003; however, both federal and California gasoline regulations began to restrict the sulfur content of gasoline in 2004. These new sulfur standards could affect NO_x emissions and confound our analysis of RVP, RFG, and CARB regulations; we therefore do not use data from this period in our analysis. The effectiveness of these sulfur programs on urban air concentrations of ozone, NO_x, and sulfur dioxide remains a topic for future research.

per year. Roughly 80% of the monitors are located in rural and suburban settings, with the remaining 20% located in urban areas.[10] The "total counties" column of table 1 indicates that the set of monitored counties grew by 1.3% per year, demonstrating that the growth in monitors came from both adding previously unmonitored counties as well as increasing the number of monitors in previously monitored counties.

[Table 1 – Approximately here]

The right-most four columns of table 1 display the number of monitored counties, by year, for each of four types of content regulation: RVP phase I (counties with RVP limits of 9.5 and 10.5 psi), RVP phase II (limits of 7.8 psi and below), federal RFG, and CARB. Counties not enumerated in any of the four columns have an RVP limit of 9.0 psi under either RVP phase I or phase II. RVP phase II is seen to begin in 1992 with the introduction of stringent RVP limits in southern ozone nonattainment areas. In 1995, approximately 30 of the monitored counties that had been observing these strict limits adopted RFG. While these counties still technically participate in the RVP program, the RVP requirements are superseded by the RFG standards. 1995 also saw about 80 monitored counties that had been observing an RVP standard of 9.0 psi adopt RFG.

Table 1 also demonstrates the introduction of CARB reformulated gasoline throughout the entire state of California in 1996. Six counties in the Los Angeles-San Diego area that had been observing the federal RFG standard switched to the CARB formulation, and the remainder of the state switched from the RVP 7.8 psi standard to CARB. Over 1996-2003, we observe monitors in 48 to 50 of these counties, out of 58 California counties in total.

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[10] The urban, suburban, and rural designations are provided by the EPA. We have spot checked 100 monitors based on their latitude and longitude data in Google Earth and confirmed this location classification based on our judgment.

B. Weather data

We control for weather in our analysis because ozone concentrations increase with temperature and sunlight. We acquired weather data measurements from the National Climatic Data Center's Cooperative Station Data (NOAA, 2008), which provide daily minimum and maximum temperatures, rain, and snowfall at more than 20,000 weather stations across the United States.[11] These weather stations are not typically located adjacent to a pollution monitor and many have missing observations. To obtain a daily weather observation at each pollution monitor, we use the following algorithm. First, we calculate the Vincenty distance of each pollution monitor to all weather stations. We then identify the ten closest weather stations to each pollution monitor, provided that each is less than 50 miles from the monitor and the elevation difference between the monitor and the station is less than 500 vertical feet. Of these stations, we identify the "primary station" as the closest station for which 50% of the pollution monitor's daily readings can be matched to the station's weather data. We then match the four climate variables for this station to the time series of ozone measurements.

Following these steps, 10.2% of the daily ozone measurements are not matched to a full set of weather variables from a primary station. We fill in these missing values by first regressing, for observations in which the primary weather station was active, the relevant weather variable for the primary station onto the same variable for the remaining nine closest stations. We use the predicted values from that regression to replace missing values. Following this step, primary station observations are still missing whenever one of the remaining nine closest stations is also missing an observation. To estimate the remaining missing values, we repeat the above step with the 8 closest stations, then the 7 closest, etc. At the end of this

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[11] Modeled weather data at an even finer spatial scale are available from the PRISM group, but only at a monthly frequency.

procedure, less than 0.1% of the remaining ozone monitor observations are still missing a matching climate observation. We drop these observations from our analysis.

To check the performance of our algorithm, we conduct the following experiment. First, we select the set of data points for which the primary weather station has an observation. We then randomly set 10% of the temperature data for this station to missing. After applying the algorithm described above to this sample, we compare the predicted temperature data to the observations we had set aside. Even for observations in which a single additional weather station was used to predict a missing temperature, the correlation coefficient between actual and predicted temperatures exceeds 0.95. Plotting the actual and predicted series against each other provides an almost perfect fit. We therefore feel confident that our algorithm provides us with a close representation of the true data generating process for missing weather observations.

C. Plots of air quality and temperature data

Figure 3 plots the daily maximum ozone concentrations in our sample, averaged across June, July, and August of each year. The data are grouped by the type of regulation employed by each county. The solid thin line plots concentrations for “control” counties that have a standard 9.0 psi summer RVP limit under RVP phase II. The dashed thin line tracks mean concentrations for counties which adopted an RVP standard of 7.8 psi or lower under RVP phase II, but were never treated with RFG or CARB. The solid thick line corresponds to counties treated with RFG but not CARB, and the dashed thick line represents CARB counties.

[Figure 3 – Approximately here]

The ozone levels across the four county types reflect the fact that gasoline regulations targeted counties with air pollution problems. RVP counties have slightly higher ozone concentrations than control counties, and RFG and CARB counties have concentrations that are substantially higher. The stringent RVP standards of 7.8 psi or lower came into effect in 1992; however, the raw data in the graph show no indication that ozone concentrations decreased in RVP counties at this time. Federal RFG began in 1995 and CARB began in 1996; the effects of these regulations cannot be clearly discerned from the graph alone. For example, while California experienced a large decrease in ozone concentrations during our sample period, it is not clear without further analysis whether these reductions were due to gasoline regulations or other factors.

Figure 4 plots the average daily maximum temperature by year for the same summer months as figure 3, broken out by the same types of counties. A comparison of these two graphs shows the strong correlation between temperature and ambient ozone concentrations. In particular, hot summers in RFG counties (1995, 1999, and 2002) and California (1994 and 1996) are associated with high ozone concentrations. This strong correlation underscores the value of using weather data to improve the efficiency of the estimators used in our formal empirical analysis.

[Figure 4 – Approximately here]

III. Empirical Strategy

Our goal is to identify the extent to which each gasoline program affects ambient ozone concentrations. Specifically, we aim to distinguish the effects of the following types of content regulation:

1. Summer RVP of 9.0 psi (some counties, 1989-1991; most ozone attainment counties, 1992 onward)

2. Summer RVP of 9.5 psi or 10.5 psi (many counties, 1989-1991)
3. Summer RVP of 7.8 psi or below (southern ozone nonattainment counties and northern opt-in counties, 1992 onward)
4. Federal RFG (severe ozone nonattainment and opt-in counties, 1995 onward)[12]
5. CARB (all California counties, 1996 onward)

Throughout our empirical discussion, we treat regulation 1, summer RVP of 9.0 psi, as a “baseline” against which the other four regulations are compared. We assess the impacts of these regulations using both a difference-in-difference (DD) method and a time series regression discontinuity (RD) design.

A. Difference-in-difference (DD)

In the DD approach, identification of the regulations’ effects comes from the year-to-year change in air quality following the introduction of a particular regulation in treated areas, compared to the contemporaneous change in control areas. We restrict our sample to the summer months of June through August, when ozone levels are at their seasonal peak and the effectiveness of gasoline regulations is most crucial. Moreover, RVP regulations are in effect only during the summer months, as are the VOC control components of RFG and CARB regulations.

We apply the DD method using a sample of observations at the monitor-day level and use both each monitor’s daily maximum reading and its daily 8 hour maximum reading as dependent variables. Our most basic DD model is given by equation (1) below, in which y_{it} denotes the value of one of these two variables as recorded at monitor i on date t. $\mathbf{Treat_{ct}}$ is a vector of four

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[12] We have carried out analyses that attempt to distinguish the effect of RFG phase II in 2000 from that of phase I. These regressions indicate no evidence of an incremental effect of RFG II and, for brevity, are omitted here.

variables indicating whether the county c in which monitor i is located is subject to one of the four possible regulatory treatments at time t (excluding the baseline RVP standard of 9.0 psi). $\boldsymbol{\alpha}$ is a four-element vector of parameters whose estimation is of primary interest.

$$(1) \qquad \ln(y_{it}) = \boldsymbol{\alpha} \cdot \mathbf{Treat_{ct}} + \mu_i + \eta_{ry} + \varepsilon_{it}$$

Equation (1) includes a set of monitor fixed effects, denoted by μ_i, that control for unobservables that cause some locations to, on average, have higher ozone concentrations than others. These fixed effects prevent the estimates of the treatment effects, $\boldsymbol{\alpha}$, from being biased upward by the fact that treated counties generally have higher levels of ozone pollution, both before and after treatment, than do control counties. Also included in (1) are fixed effects η_{ry} for the interaction of the four U.S. census regions r with each year y. These interactions control for unobserved year-to-year shocks that are common to both treated and untreated monitors within each census region. Finally, ε_{it} represents an unobserved disturbance.

The identification assumption underlying (1) is that county-specific unobserved factors affecting ozone concentrations are constant over time. Formally, identification of $\boldsymbol{\alpha}$ requires that $E[\mathbf{Treat_{ct}} \cdot \varepsilon_{it} \mid \mu_i, \eta_{ry}] = 0$. This assumption may not hold, however, if air quality in treated counties has a long-term trend that differs from the trend in control counties. Treated counties are generally those that are in non-attainment for ozone and may experience forces that cause their ozone concentrations to increase or decrease over time relative to control counties. Economic activity may grow more quickly in treated areas, which tend to be relatively urban, putting upward pressure on ozone. However, these areas may also be undertaking pollution abatement actions that could result in a downward trend, relative to control counties. In either case, such a differential trend in treated vs. control counties will bias the estimate of $\boldsymbol{\alpha}$: the first case will create upward bias, while the second will create downward bias.

To both improve the precision of our estimates and control for factors that affect treated and control counties differentially over time, we augment (1) with additional variables to form specification (2) below:

$$\ln(y_{it}) = \boldsymbol{\alpha}\cdot \mathbf{Treat_{ct}} + \boldsymbol{\beta}\cdot\mathbf{W_{it}} + \gamma_r\cdot\mathbf{D_t} + \delta\cdot I_{ct} + \boldsymbol{\theta}\cdot\mathbf{Trend_{rct}} + \mu_i + \eta_{ry} + \varepsilon_{it} \tag{2}$$

In (2), the variables $\mathbf{W_{it}}$ control for monitor-specific weather shocks and include a flexible polynomial in temperature and precipitation, as well as interactions of these variables with day-of-year and day-of-week.[13] $\mathbf{D_t}$ denotes a vector consisting of six dummy variables for day-of-week and a day-of-year variable. The coefficients γ_r on $\mathbf{D_t}$ are census region-specific. I_{ct} denotes county-level total annual personal income (Bureau of Economic Analysis, 2008). Finally, the set of variables denoted by $\mathbf{Trend_{rct}}$ are linear time trends that are specific to treated and control counties within each census region. That is, counties that are treated with RFG in census region 1 are given a trend that is distinct from region 1 counties that are treated with RVP or not treated at all.[14] These trends are included in the specification to attempt to distinguish the impacts of gasoline regulations from long-run trends driven by unobservables.[15] In some of our specifications, these trends include quadratic terms.

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[13] Specifically, $\mathbf{W_{it}}$ includes cubic polynomials in maximum and minimum temperature (T_{max} and T_{min}), the interaction of T_{max} and T_{min}, quadratics in rainfall and snowfall, the interaction of rainfall with T_{max}, one-day lags of T_{max} and T_{min}, T_{max} interacted with lagged T_{max}, and T_{max} interacted with lagged T_{min}. $\mathbf{W_{it}}$ also interacts all of these variables with a day-of-year variable to allow weather effects to vary over the summer, and interacts T_{max}, T_{min}, rainfall, and snowfall with day-of-week dummies to allow for variations in ozone formation on weekdays and weekends.

[14] Moreover, counties that are initially treated with RVP and then subsequently treated with RFG receive a time trend that is distinct from those of counties that were treated with only one of RVP or RFG. CARB counties, as well as counties in the Los Angeles area that had federal RFG in 1995 and CARB from 1996 onwards, also receive their own trends.

[15] We have also attempted to control for unobserved time-varying factors by estimating a version of (2) that includes dummy variables for each county's attainment status for ozone and five other "criteria" pollutants: particulate matter, NO_x, sulfur dioxide, carbon monoxide, and lead. Including these variables has only a negligible impact on the estimated treatment effects, reflecting the fact that very few treated counties change their ozone attainment status during the sample period.

The identification assumption of the augmented DD model (2) is that unobserved factors are not correlated with treatment, conditional on the covariates; that is, $E[\mathbf{Treat_{ct}} \cdot \varepsilon_{it} \mid \mathbf{W_{it}}, \mathbf{D_t}, I_{ct}, \mathbf{Trend_{rct}}, \mu_i, \eta_{ry}] = 0$. This identification assumption, while more relaxed than that of (1), may nonetheless be invalid if unobserved factors exist that affect ozone concentrations in a way that is non-linear over time and not captured by any of $\mathbf{W_{it}}$, $\mathbf{D_t}$, or I_{ct}.

For inference, we allow the unobserved disturbance ε_{it} to be correlated across all observations within the same state and year, addressing both serial correlation, per Marianne Bertrand, Esther Duflo and Sendhil Mullainathan (2004), and within-state cross-sectional correlation. The standard errors we report therefore use a robust variance estimator that is clustered on each state-year combination (Manuel Arellano 1987, Jeffrey M. Wooldridge 2003).[16]

B. Regression discontinuity (RD) design

In the RD approach, identification of the regulations' effects comes from the change in ozone concentration within a narrow window around the phase-in of each regulation. We are able to focus on a short time period because: (1) imposition of a gasoline standard affects all cars simultaneously, implying that the standard will cause a step change in emissions almost immediately after implementation; and (2) ozone decomposes overnight, meaning that daily maximum ozone concentrations will respond quickly to changes in emissions. This approach permits an identification assumption that is more relaxed than that of the DD model. While identification of the DD equation (2) requires that unobserved variables affecting ozone

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[16] We investigated whether within-monitor, cross-year correlation is important by regressing the residuals from equation (2) on the within-year residuals and a set of residuals lagged by approximately one year. When this set of one year lagged residuals includes only the 365 day lag, the point estimate of this lag is only 0.003 and is not statistically significant. When additional lags ranging from 365 to 390 days are also included, some are statistically significant, but the magnitudes of the point estimates are always lower than 0.07. We therefore believe that clustering on state-year yields an accurate estimate of our estimates' standard errors. We repeated this exercise for our RD estimates and obtained similar results.

concentrations do so only through a linear time trend, the RD model permits unobserved factors to act non-linearly over time, so long as they are not discontinuous when gasoline regulations phase-in (Jinyong Hahn, Petra Todd, and Wilbert Van der Klaauw 2001).[17]

Our implementation of the RD design is similar to that of Davis (2008). We estimate equation (3) below, which is more flexible than the DD model (2) in a number of ways. In (3), the treatment effects $\boldsymbol{\alpha}_{\mathbf{i}}$ and the coefficients $\boldsymbol{\beta}_{\mathbf{i}}$ on the weather variables are monitor-specific.[18] In addition, the linear time trend in (2) is replaced in (3) with $f_i(Date_t)$, an eighth-order Chebychev polynomial in time that is also monitor-specific.[19] We therefore estimate (3) one monitor at a time as a time series regression. This monitor-specific approach permits considerable flexibility in the manner in which both observed and unobserved factors can influence ozone concentrations while simultaneously allowing us to evaluate spatial heterogeneity in the treatment effects.

$$\ln(y_{it}) = \boldsymbol{\alpha}_{\mathbf{i}} \cdot \mathbf{Treat}_{\mathbf{ct}} + \boldsymbol{\beta}_{\mathbf{i}} \cdot \mathbf{W}_{\mathbf{it}} + f_i(Date_t) + \mu_i + \varepsilon_{it} \tag{3}$$

We estimate (3) using data from all seasons of the year so that observations near the regulatory transitions that occur in the spring and fall are included in the sample. We restrict the sample to monitors that deliver valid daily readings for more than 75% of each calendar quarter for 75% of all possible quarters. This sample therefore represents a set of monitors that

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[17] The discontinuity in our approach is across time, rather than a more traditional cross-sectional discontinuity at a treatment threshold. We do not use a cross-sectional threshold, such as the ozone concentration that triggers a non-attainment designation by the EPA, because such an approach would not address the factors we seek to control for in the RD: other regulatory actions that are brought about by a non-attainment designation. Moreover, this approach would be of no help in evaluating the impact of CARB regulation, which affected every county in California regardless of its pre-treatment ozone level.

[18] In the RD model (3) we expand the number of weather variables by interacting them with seasonal dummies, and we also include stand-alone month dummies.

[19] Use of a 7th, 9th, or 10th order polynomial yields results very similar to those reported in this paper. Shortening the sample window from 1989-2003 to 1989-1999 also does not substantially alter the results.

consistently record concentrations year-round and is smaller than the set of summer monitors used in the DD analysis.[20]

To identify the effects of RVP phase II, we focus on counties that were not also treated with RFG or CARB in a year subsequent to their RVP treatment. With only three years of pre-treatment observations in most RVP counties, we ensure that there are a sufficient number of pre-treatment observations by enforcing that all monitors be active for 75% of the days in 75% of the quarters prior to treatment. We model RVP II regulations with a treatment dummy that is active only during the summer VOC control periods, reflecting the seasonality with which these regulations are applied. Specifically, the element of $\mathbf{Treat}_{\mathbf{ct}}$ in (3) corresponding to RVP phase II takes on a value of one if retail gasoline stations in county *c* are required to stock and sell RVP-limited gasoline on date *t*.

In evaluating the impact of federal RFG, we control for the fact that some RFG counties, primarily those in the South, were also treated with RVP II in 1992 prior to initiating RFG in 1995. We therefore include in our RFG RD regressions all elements of $\mathbf{Treat}_{\mathbf{ct}}$ that are applicable to each county.[21] As with RVP, we model the RFG treatment dummy as active during only the summer VOC control periods.[22]

The regulatory discontinuity at the start and end of each summer's VOC control period—for both RVP and RFG—is not sharp. Gasoline at retail stations and in the gas tanks of vehicles does completely turn over in a single day. We therefore allow for a 30 day linear phase-in of the

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[20] When the DD specification (2) is estimated using the same sample of monitors used to estimate the RD specification (3), results are similar to the DD results shown in table 3.

[21] Omitting the RVP treatment dummy for those counties that were also treated with RVP has only a minor impact on the reported results.

[22] We have also examined RD regressions in which the RFG treatment effect is specified as a year-round impact, reflecting the potential for some of the year-round toxic emission reductions associated with RFG to reduce ozone formation. Consistent with our seasonal RFG results described below, we do not find evidence of a significant impact of RFG with this specification. Specifically, the average treatment effect across monitors is -0.014, and we find a statistically significant decrease in ozone (at the 5% level) at only 3 of 41 monitors. The significance of the results at these three monitors is not robust across alternative polynomial specifications.

treatment effect between the date that refiners are required to produce the regulated gasoline and the date that gas stations are required to stock and sell it. At the end of the summer VOC control season we allow a 30 day linear phase-out as conventional gasoline returns to retail stations and vehicles' gas tanks.

In our RD estimation of the effect of CARB regulation, we model CARB as a year-round treatment that becomes effective on the date of its introduction: 1 March, 1996. While CARB does include a seasonal RVP limit, CARB's restrictions on the content of olefins and aromatic hydrocarbons—both highly reactive in forming ozone—are year-round.[23] We allow for a linear ramp in the element of $\mathbf{Treat_{ct}}$ corresponding to CARB over the 30 days prior to 1 March, 1996.

Our RD estimation of CARB's effectiveness is complicated by the facts that the entire state of California was subject to an RVP standard of 7.8 psi beginning in 1992 and that six counties in southern California—Los Angeles, Orange, Riverside, San Bernardino, San Diego, and Ventura—were also treated with federal RFG in 1995. In estimating (3), we therefore include in our CARB regressions all elements of $\mathbf{Treat_{ct}}$ that are applicable to each county. For example, in modeling ozone concentrations in Los Angeles, we control for potential impacts from RVP during the summers of 1992-1994 and RFG during the summer of 1995.[24]

In all RD specifications, we allow each monitor's time-varying unobserved disturbance ε_{it} to be correlated across all observations within each year-season. Thus, we estimate standard errors of our parameter estimates using a clustered robust variance estimator (Arellano 1987, Wooldridge 2003).

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[23] Even though ground-level ozone concentrations are relatively low in the winter, CARB enforces its olefin and aromatic hydrocarbon content limits in all months of the year because these compounds are toxic and therefore cause health problems independently of their ability to form ozone (CARB, 2009). We have examined RD specifications in which we allow for both a year-round and summer-specific impact of CARB. Estimates from these specifications indicate that, while there is a strong year-round impact of CARB per the results discussed below, there is no evidence of an additional seasonal effect.

[24] Omitting the RVP and RFG treatment dummies yields slightly stronger CARB treatment effects than those reported here.

IV. Results

A. Difference-in-difference (DD) results

Plots of yearly residuals

Figure 5 depicts the time path of ozone concentrations in counties under different forms of regulation while removing the noise associated with weather shocks. Each panel plots the residuals of a regression of daily maximum ozone concentrations on the weather, day-of-week, and day-of-year variables $\mathbf{W}_{\mathbf{it}}$ and $\mathbf{D}_{\mathbf{t}}$, as well as monitor fixed effects. These residuals are averaged across monitors and dates within each year and regulation type, and offer insight into the estimates of the DD model.

[Figure 5 – Approximately here]

Panel (a) compares the ozone residuals in counties treated with a stringent RVP phase II standard of 7.8 psi or lower (and never treated with RFG or CARB) to baseline counties with an RVP limit of 9.0 psi. These two sets of residuals track each other very closely throughout the sample period: they are typically no farther apart than 0.001 ppm, relative to average concentrations of about 0.06 ppm. The introduction of 7.8 psi RVP gasoline, which occurred in 1992 in most treated counties, does not appear to have substantially affected summertime ozone concentrations.

Panel (b) suggests that the introduction of federal RFG in 1995 may have caused modest reductions in ozone pollution. Prior to 1995, the residual ozone concentrations for RFG counties are higher than those of baseline counties, whereas this pattern is reversed from 1995 onwards. It is possible, however, that this shift may have been part of a trend in which ozone concentrations in RFG counties, relative to baseline counties, gradually decreased over 1992-1996. Similarly, panel (c) indicates a substantial decrease in ozone concentrations in California counties around the time when CARB gasoline was introduced in 1996. Again, it is difficult to discern from this graph alone whether this decrease can be attributed to CARB gasoline or to other factors acting over a multi-year time span. The interpretation of panel (c) is also made difficult by the fact that the heavily monitored counties in the Los Angeles and San Diego areas were treated with federal RFG in 1995 before adopting CARB standards.

DD Estimation results

The DD estimates of the effect of gasoline content regulations on summer ozone concentrations are given in table 2. The top four rows indicate the estimated effect of RVP phase I (RVPs of 9.5 and 10.5 psi), RVP phase II (RVPs of 7.8 psi and lower), federal RFG, and CARB standards on the logarithm of ozone concentration, all relative to the baseline RVP standard of 9.0 psi. Column I displays the results of estimating specification (1), which includes monitor fixed effects μ_i and region-year effects η_{ry} but no additional controls, and uses the logarithm of the daily maximum ozone concentration as the dependent variable. Neither RVP phase I nor RVP phase II is estimated to have a significant impact on ozone. While both point estimates do have the correct sign (the high phase I RVPs increase ozone relative to the baseline, while the low phase II RVPs decrease it), they are small in magnitude and statistically insignificant. We estimate that imposition of federal RFG is associated with a modest decline in log(ozone) of -0.029, statistically significant at the 1% level and statistically distinct from the effect of RVP II (the p-value of the F-test for equality of these coefficients is 0.078). Only CARB gasoline is estimated to have a large impact on ozone concentrations: the implementation of CARB standards is associated with a decrease in log(ozone) of -0.095 (equivalent to a 9.1%

decrease in absolute ozone concentration) that is statistically significant at the 1% level. The data reject equality of the CARB coefficient with that of either RVP II or RFG (the p-values are less than 0.001 in both cases).

[Table 2 – Approximately here]

Columns II through V of table 2 progressively add control variables to the specification, moving from an estimate of equation (1) in column I to equation (2) in columns IV and V. Column II includes variables for weather ($\mathbf{W_{it}}$) as well as day-of-week and day-of-year ($\mathbf{D_t}$). These variables have little effect on the point estimates, though the standard errors decrease slightly and the regression's R^2 increases substantially from 0.02 to 0.26. Nearly all of this increase is due to the inclusion of the weather variables.

Column III controls for county-level annual personal income. The estimated coefficient on income is negative and statistically significant, consistent with increases in pollution abatement in high-growth counties. Relative to column II, controlling for income causes modest declines in the estimated effects of RVP II, RFG, and CARB regulations. In this specification, we no longer reject equality of the coefficients on RVP II and RFG (p-value: 0.328).

Columns IV and V add linear and quadratic regulation-specific time trends to the specification. These additions result in minor changes to the estimated impact of RVP and slightly attenuate CARB's impact. The estimated effect of RFG, however, varies substantially across these specifications: with a linear time trend we estimate that RFG reduces log(ozone) by a statistically significant 0.036, but with a quadratic trend this effect is only -0.019 and not statistically significant. The equality of the RVP and RFG coefficients is rejected in column IV (p-value: 0.046) but not column V (p-value: 0.623).[25] The sensitivity of these results to changes in the specification of the time trends underscores the importance of carefully controlling for time-varying unobservables, motivating our more flexible RD design.

Columns VI through VIII of table 2 repeat the specifications in columns I through V but use the logarithm of the daily 8 hour ozone concentration as the dependent variable rather than the daily maximum. These estimated effects vary little from those presented in columns I through V.

Table 3 presents results that are broken out by whether each monitor is in an urban, suburban, or rural area. The specification used in columns II, IV, and VI is exactly that given by (2), while columns I, III, and V do not include the time trends. Again, little evidence is found of an impact of RVP regulations on ozone concentrations, with the exception of a marginally significant estimate in suburban areas when linear time trends are included. The effect of RFG is estimated to be strongest in suburban areas; in urban and rural areas RFG is estimated to only have a marginally statistically significant effect, reducing ozone concentrations by 0.4 to 3.1% depending on the specification. CARB gasoline has significant effects in both urban and suburban areas of about 7% and 10%, respectively. CARB's effectiveness in rural areas is weaker: it is estimated to significantly decrease ozone concentrations by 4% in specification V, but the inclusion of time trends in specification VI yields a small, insignificant effect of 1.7%.

[Table 3 – Approximately here]

The diminished effectiveness of both federal RFG and CARB gasoline in rural areas likely reflects the Leontief nature of the ozone production function. Because rural areas tend to

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[25] The equality of the effects of RVP II and CARB is rejected across all specifications at the 1% level. The equality of RFG and CARB is rejected at at least the 5% level in all specifications save column IV, in which the p-value is 0.144.

be NO_x-limited, gasoline regulations, which primarily affect VOC emissions, may be relatively ineffective in these regions.

Table 4 presents results obtained by estimating the same pooled specifications presented in table 2, but using a dataset consisting of only those monitors that report observations during every year of the 1989-2003 sample period. These results generally report treatment effects that are more negative than those reported in table 2, indicating that the full-time monitors located in treated areas experienced a decrease in ozone concentrations relative to part-time monitors. This outcome could result from the placement of new monitors in areas that are experiencing increases in ozone pollution.

[Table 4 – Approximately here]

Point estimates for the effect of CARB gasoline on log(ozone) in table 4 range from -0.118 to -0.148, and estimates for RFG range from -0.018 to -0.055. F-tests reject equality of the RFG and CARB coefficients at the 1% level across all specifications. The effect of RVP phase II is now statistically significant in columns IV and V, in which time trends are included, with estimated magnitudes of -0.025 and -0.022, respectively. However, the effects of RVP phase I standards weaker than the 9.0 psi baseline are also estimated to be negative in these specifications, with magnitudes of -0.019 and -0.015 that are not statistically significant. Thus, the estimated effect of a stringent RVP phase II standard is not significantly different from that of a lax RVP phase I standard, suggesting again that RVP regulations are not effective in reducing ozone concentrations.

Overall, the results presented in this section provide evidence that CARB gasoline substantially decreases ozone concentrations and RVP gasoline does not. The evidence for federal RFG is mixed, with some specifications indicating a modest effect and others not. The magnitudes of the estimated effects are somewhat sensitive to the inclusion of time trends and the choice of monitors included in the sample. The regression discontinuity design addresses these issues by allowing for time-varying unobservables and substantial monitor-level heterogeneity.

B. Regression discontinuity results: RVP and RFG

The treatment effects we estimate using the RD design are monitor-specific. Figure 6 plots daily ozone concentrations for monitors that are characteristic of our results for RVP phase II and RFG gasoline. The fitted line for each monitor is the time series of predicted values of the treatment effect and polynomial time trend $(\boldsymbol{\alpha}_{\mathbf{i}}\cdot \mathbf{Treat}_{\mathbf{ct}} + f_i(Date_t))$ obtained from estimating equation (3), centered so that its mean value is zero. The points plotted are the sum of this fitted line with the residuals from estimating (3). These plots therefore illustrate our RD specification for RVP and RFG: an abrupt change in the seasonality of the residuals and the fitted line in the year a regulation becomes active is indicative of a significant effect of the regulation on ground-level ozone pollution.

[Figure 6 – Approximately here]

Panel (a) depicts the RD result for pollution monitor 3007 in Madison County, Illinois (just east of St. Louis, Missouri) which was treated with an RVP phase II standard of 7.2 psi in 1995. A shift in neither the residuals nor the fitted line is apparent in 1995. Accordingly, the RD estimate of the effect of RVP on log(ozone) for this monitor is +0.004 and statistically insignificant.

The dotted line in figure 7, panel (a) displays a kernel-smoothed cross-monitor distribution of the RD estimates of RVP phase II’s effect on the logarithm of ozone concentration. This plot also displays our estimated effects for RFG (dashed) and CARB (solid). The mean of the RVP distribution, across 46 monitors, is -0.0001, and monitor 3007 in Madison County, Illinois, depicted in figure 6, lies nearest this mean. Table A4 in the appendix lists the county-specific RD estimates that make up this distribution, as well as the distributions of RFG's and CARB's impacts. On average, we find that imposition of RVP does not cause a significant reduction in ozone concentrations, consistent with the DD results discussed above.[26]

The distribution of the RFG estimates depicted in panel (a) of figure 7 is shifted slightly to the left relative to that of RVP. Its mean is -0.021, which is smaller in magnitude than most of the DD estimates of RFG's impact. This distribution includes results from 41 monitors, 13 of which are found to have reductions in ozone that are statistically significant at the 5% level. These monitors are almost all located on the eastern seaboard between Wilmington, Delaware and Long Island, New York (monitor-specific point estimates may be found in table A4 in the appendix). Figure 6b plots residual ozone concentrations and a fitted line for monitor 1001 in Camden County, New Jersey: ozone at this location clearly exhibits substantial seasonality following the 1995 introduction of RFG.

Why might ozone concentrations have decreased in summer 1995 (and the following summers) along the eastern seaboard, but not in other RFG locations? This pattern of effects seems consistent with the introduction of substantial NO_x emission controls in Delaware, New Jersey, New York, and Pennsylvania (the state immediately upwind) in 1994 and 1995. Table 5 lists the frequency of installation of NO_x control equipment at electric generating units by year for several groups of states. Delaware, Pennsylvania, New Jersey, and New York together account for about one-third of all electric generation NO_x control installations in the U.S. during

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[26] The only monitors for which we find negative and statistically significant impacts of RVP are located in southern Florida, where Hurricane Andrew struck in August 1992. That summer was the first in which RVP was in effect. The severe economic disruption caused by the hurricane seems likely to explain these observed decreases in ozone.

1994 and 1995 and were the only RFG control areas to experience such a surge in activity at this time.[27]

[Table 5 – Approximately here]

We test this proposition by incorporating table 5's data on NO_x control installations into the RD regressions for monitors in Delaware, New Jersey, New York, and Pennsylvania. We create a $TreatNOx_t$ variable that is a cumulative count of the NO_x installations that have occurred in DE, NJ, NY, and PA by date t. We set $TreatNOx_t$ to zero for all months outside of the ozone season to reflect the seasonal use of NO_x control equipment.[28] When $TreatNOx_t$ is included in the regressions, the average coefficient on the RFG treatment dummy falls from -0.072 to -0.011 over the 17 monitors in the region, and the average t-statistic falls from -2.29 to -0.307 (monitor-by-monitor results are given in table A5 in the appendix). In contrast, we find a significant impact from NO_x control: the average impact of 100 additional NO_x installations on log(ozone) is estimated to be -0.050 with an average t-statistic of -1.88.[29] These results are evidence that the summer ozone reductions along the eastern seaboard were driven by local and upwind reductions in NO_x emissions rather than an effect of the RFG program.

At locations outside of Delaware, New Jersey, and New York, the average estimated impact of RFG is +0.016. The estimated impact at monitor 47 in Harris County, Texas (Houston) nearly matches this mean, and figure 6c plots ozone residuals and a fitted trend at this location. A shift in neither the residuals nor the fitted line is apparent in 1995, when RFG was imposed.

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[27] There were also a substantial number of installations in North Carolina; however, this state is not an RFG area nor is it usually upwind of any such areas.

[28] Electric generators deactivate NO_x control equipment outside of the ozone season because it imposes high operating costs. The EPA-designated ozone season for NO_x control is 1 May through 30 September.

[29] Separate identification of the NO_x control effect from the RFG effect comes from the fact that the RFG treatment is constant across all summers after 1995, whereas the NO_x treatment increases in magnitude each summer as generators make more investments in pollution control equipment. In particular, a second wave of NO_x abatement installations in 2001 and 2002 appears to have contributed to additional reductions in summer ozone that cannot be explained by RFG.

Finally, we note that the lack of evidence tying RFG to reductions in ozone is not caused by an absence of VOC-limited areas in our sample of monitors. A number of monitors are located in urban areas such as Chicago, Houston, Philadelphia, and New York that are thought to be VOC-limited (Sillman 1999, Blanchard 2000, 2001, and Reynolds *et al.* 2004). At none of these monitors do we find evidence of a substantial decrease in ozone associated with RFG. In fact, the eastern seaboard monitors at which we find ozone reductions are suburban locations that are likely to be NO_x-limited (Charles L. Blanchard 2001), further supporting the claim that these reductions were driven by NO_x emission abatement rather than RFG. Overall, we conclude that RFG, like RVP, does not substantially reduce ozone pollution.

C. Regression discontinuity results: CARB

[Figure 7 – Approximately here]

[Figure 8 – Approximately here]

Figure 8 illustrates our RD strategy for identifying the impact of CARB regulation. As with figure 6, the points plotted in each panel are residual ozone concentrations recorded at the indicated pollution monitor. Because California was treated not only with CARB in 1996, but also RVP (7.8 psi) in 1992 and RFG in 1995 (in the Los Angeles and San Diego area), the fitted line includes treatment dummies for all three types of regulation. Panel (a) depicts residual ozone concentrations at monitor 1201 in coastal Los Angeles County, California, while panel (b) displays the residuals for monitor 1701, located in the interior of Los Angeles County just east (and downwind) of the intersection of the San Bernardino Freeway (I-10) with the Orange Freeway (SR-57). These two locations are indicated on a map in figure 9, panel (a).

While the coastal monitor in panel (a) of figure 8 indicates little evidence of an impact of CARB on ozone, the residual concentrations shown for the inland monitor 1701 in panel (b) reveal an abrupt, substantial reduction in ozone concentrations in early 1996. These plots reflect our RD estimates of CARB's effect at each monitor. For the coastal monitor, our point estimate is that CARB reduced ozone concentrations by 10.2%; however, this result is not statistically distinct from zero (the t-statistic is -0.986). At the inland monitoring location, in contrast, we find that CARB reduced ozone concentrations by 35.2%, statistically significant at the 1% level.

The impact of CARB gasoline on ozone is therefore spatially heterogeneous. The spatial distribution of our monitor-specific estimates is indicated on the maps of Los Angeles and California in figure 9, panels (a) and (b), respectively. Specific point estimates and standard errors for each monitor are given in table A4 in the appendix.[30] Nearly all of the statistically and economically significant effects are found in the inland Los Angeles and San Diego areas that feature high temperatures, dense populations, high baseline ozone levels, and VOC-limited ozone formation conditions. Areas of the state that do not possess all of these features, such as the San Francisco Bay area or the rural Central Valley, do not appear to benefit substantially from CARB regulation. This pattern of outcomes is consistent with the science of ozone formation. Reductions in the content of ozone-forming VOCs in gasoline will be most effective in areas where gasoline-emitted VOCs drive substantial ozone formation: these areas will be hot, densely populated, and VOC-limited.[31]

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[30] We have also used a local linear regression estimation strategy, which first strips out seasonal and weather effects and then evaluates the CARB treatment effect using regressions on data limited to a specified bandwidth around the March 1996 treatment date. Using bandwidths of 365 and 180 days on both sides of this date, we find results that are very similar to those presented in table A4. At a 90 day bandwidth, point estimates are similar to those in table A4, though standard errors increase so that most estimates are not statistically significant at conventional levels.

[31] To the best of our knowledge, and after extensive checking with CARB, no new VOC or NO_x regulations were coincident with CARB in 1996. There were zero NO_x emission control installations at California power plants in 1996 (table 5). Of the 6 installations in California in 1995, 4 were in Los Angeles County and 2 were in Ventura County. 5 of these 6 were installed by June, nine months before the onset of CARB and the discontinuity in the

[Figure 9 – Approximately here]

The kernel density plot of figure 7, panel (a) indicates that the distribution of CARB's effect on ozone is shifted to the left relative to that of RFG or RVP: the mean estimated effect of CARB on log(ozone) is -0.060. Moreover, the CARB distribution features a large left tail of substantial, statistically significant impacts that is not present in the other distributions. This left tail is drawn almost entirely from monitors in the Los Angeles and San Diego area.

One difficulty in interpreting figure 7, panel (a) is that the southern California monitors driving the left tail of CARB results are not represented in the RVP and RFG results. Even though urban areas that are thought to be VOC-limited are represented in the RVP and RFG regressions, it may be that unique features of southern California drive the differences in estimated regulatory effectiveness rather than the regulations themselves. Panel (b) of figure 7 addresses this issue by plotting distributions of the estimated impacts of RVP, RFG, and CARB in southern California.[32] The distribution of RFG's estimated impacts has a mean near zero at +0.021, and the RVP distribution is centered slightly to the right at +0.055. These last results demonstrate that RVP and RFG regulations were ineffective at the same locations for which CARB regulation significantly reduced ozone concentrations.[33]

Overall, the estimates obtained from the RD strategy reaffirm the findings from the DD model and indicate the presence of spatially heterogeneous effects. We find no evidence that

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ozone data. Even putting aside this mis-timing, these installations cannot explain the March 1996 decrease in ozone in San Diego county, well to the South (and not downwind of) Los Angeles. Finally, Meredith Fowlie, Stephen P. Holland and Erin T. Mansur's (2009) study of the NOx cap-and-trade market in California indicates that NOx emissions in the Los Angeles basin did not substantially decrease in 1996 (figures 2 and 3 in their paper).

[32] The seasonal RVP and RFG treatment effects shown in panel (b) of figure 8 were estimated jointly with the CARB treatment effect per the discussion in section IIIB. We have also estimated these effects using data that exclude data from 1996 onward, when CARB was in effect. While doing so results in noisier estimates due to the diminished sample size, we still find no evidence that either RVP or RFG was effective in southern California.

[33] Moreover, the spatial distribution of the estimated RVP and RFG effects are not systematic, as is the case with CARB. That is, the negative part of the RVP and RFG distributions are not concentrated at the inland southern California monitors, as would be expected if RVP and RFG were effective in reducing ozone concentrations. We also find no evidence that RFG has a year-round impact in southern California.

federal RVP or RFG standards are effective in reducing ozone pollution. While we find decreases in summer ozone in Delaware, New Jersey, and New York that are coincident with the onset of RFG regulation in these locations, these impacts are better explained by local and upwind NO_x emissions controls. In California, however, the adoption of CARB standards in 1996 caused a large, significant decrease in ozone concentrations in the densely populated and VOC-sensitive southern part of the state. RVP and RFG controls that were introduced in this region prior to adoption of CARB did not result in reductions in ozone.

V. The importance of refiners' behavioral response

The likely explanation for the failure of RVP and RFG regulations to reduce ambient ozone concentrations centers on the flexibility granted to refiners in meeting the regulations' VOC emissions reduction standards. VOCs include a large number of compounds, and while RVP and RFG standards cap the overall rate of VOC emissions from gasoline, they allow refiners to choose which particular VOCs to remove.

Refiners meet RVP and RFG requirements primarily by removing the VOC butane from their gasoline, as noted by EPA rulemaking.[34] Butane is a light, highly volatile compound that refiners typically blend into conventional gasoline to increase its octane rating. Reducing the amount of butane that is blended is the most cost-effective avenue available to refiners to meet VOC emission reduction standards. However, butane is not highly reactive in forming ozone (CARB, 2007). Thus, even though gasoline that meets RVP or RFG standards emits a lower

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[34] See in particular the final RVP rule in 40 CFR Part 80 (also in the federal register at 54 FR 11868) and the EPA's regulatory impact analysis for RFG (1993).

volume of VOCs than does conventional gasoline, this emissions reduction does not translate into reductions in ground-level ozone.[35]

Evidence from both the butane market and air quality measurements indicates that the VOC emission reduction mandates of RVP and RFG substantially reduced the use of butane in summer gasoline. Lidderdale (1999) evaluates the impact of RVP and RFG on refining operations and finds that refiners' summer blending of butane substantially decreased following the imposition of these standards. In figure 10, we replicate one of Lidderdale's graphs using data from the Energy Information Administration on refinery net production of butane. This plot indicates that butane production becomes highly seasonal beginning in 1989, when RVP standards are first implemented. This seasonality is accentuated following the imposition of RVP phase II in 1992 and more modestly following RFG in 1995, which affected fewer areas. Under RVP and RFG, refinery net production of butane is high in the summer, meaning that refiners produce butane as a refinery output rather than blend it into gasoline. The negative net production of butane in the winter indicates that the butane produced in the summer is stored and then blended back into gasoline during the winter when the VOC standards are not in effect.

[Figure 10 – Approximately here]

The decrease in summer gasoline's butane content is also observable in ambient air quality measurements. Ben H. Lee *et al.* (2006) measure air concentrations of a variety of

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[35] This explanation behind RVP's ineffectiveness prompts the question of why RVP regulations, as well as the VOC reduction mandates of federal RFG, do not distinguish between high and low-reactivity VOCs. With regard to RVP, conversations with EPA personnel have indicated that removing butane from gasoline can, in principle, lead to reduced emissions of other, more reactive compounds (when butane evaporates from gasoline, it carries other VOCs with it), and that this belief likely played a role in EPA forecasts of ozone benefits from RVP. However, the only study we have identified that directly examines this effect (Robert M. Reuter *et al.* 1992) found highly inconclusive results regarding the difference between the ozone forming potential of emissions from RVP and non-RVP gasoline, leading us to conclude that the science regarding the magnitude of this effect is unsettled. As for RFG, according to its final Regulatory Impact Analysis (EPA 1993), the EPA claims that the Clean Air Act Amendments (CAAAs) of 1990 forbid it from using reactivity-weighting when setting VOC emission reduction standards because the law states that these standards must be set on a "mass basis." The Regulatory Impact Analysis further notes that, while early versions of the CAAAs did not include this "mass basis" statement, the wording was added during the meetings of a conference committee to reconcile differences between the House and Senate versions of the bill.

anthropogenic VOCs in Massachusetts over 1992-1996. They find that, unlike other VOCs, concentrations of butane dip during the summer. Figure 11 is a reproduction from their paper and clearly indicates that, while the concentrations of pentane and hexane—VOCs similar to butane—peak during the summer, summer concentrations of butane are low relative to its concentrations during the winter. The authors attribute this result to the RVP and RFG standards in place in Massachusetts during the years these measurements were taken.

[Figure 11 – Approximately here]

In contrast to RVP and RFG, CARB regulations include restrictions on specific VOCs that are highly reactive in forming ozone. CARB gasoline imposes content limits on classes of compounds called olefins and aromatic hydrocarbons that are three to ten times more reactive than butanes. CARB therefore denies refiners the flexibility to choose which VOCs to remove from gasoline and forces them to target components that can significantly impact ozone formation, even though these components are more expensive to remove. Thus, we observe substantial air quality improvements following the imposition of CARB gasoline.

VI. Conclusions

This paper examines the effectiveness of three types of gasoline content regulations: federal Reid vapor pressure (RVP) standards, federal reformulated gasoline (RFG), and California reformulated gasoline (CARB). Using ground-level ozone concentration data from the EPA's monitoring network, we find that the imposition of CARB standards substantially reduces ozone pollution, particularly in areas with the most severe *ex ante* ozone problems. There is no evidence, however, that the RVP or RFG regulations result in significant ozone reductions, even at the same locations at which CARB was effective.[36] RVP and RFG nonetheless impart substantial costs on consumers, since the entire country is subject to one of them during the summer months. Given U.S. non-California 2008 summer gasoline consumption of 47 billion gallons and a $0.01 - $0.015 per gallon price effect estimated in Brown *et al.* (2008), the VOC standards imposed by these regulations increase U.S. annual gasoline expenditures by $524 - $784 million.[37]

The ineffectiveness of RVP and RFG in reducing ozone can be explained by the cost-minimizing response of refiners to the regulations. Because these standards grant refiners broad flexibility in deciding which specific VOCs to remove from their gasoline, they are able to reduce the content of a particular VOC, butane, which is relatively cheap to remove but is not prone to forming ozone. In contrast, CARB gasoline mandates reductions in concentrations of highly reactive VOCs and yields substantial reductions in ground-level ozone pollution. These outcomes highlight a potential pitfall of flexible environmental regulations: while allowing flexible responses to standards can mitigate abatement costs, flexibility can also reduce the desired environmental benefits.

The air quality improvement driven by CARB gasoline is substantial in the densely populated southern part of the state. The introduction of CARB in 1996 reduced ozone concentrations by more than 20% at some monitoring locations. Our regression discontinuity estimates indicate that the average ozone reduction across monitors in the severe Los Angeles – San Diego ozone non-attainment area is 15.6%. Other California counties that are cooler or less

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[36] RFG also has elements directed at reducing carbon monoxide and carcinogenic toxics emissions. This paper does not evaluate these potential benefits. RVP is directed solely at reducing VOC emissions.

[37] The $0.01 - $0.015 per gallon effect estimated in Brown *et al*. (2008) applies specifically to the price differential between RVP 7.8 psi gasoline and RVP 9.0 psi gasoline. The above calculation assumes that relaxing the summer VOC emission standards of RFG and relaxing the federal “baseline” 9.0 psi standard would have similar price effects. The 47 billion gallon consumption figure was sourced from the EIA (http://tonto.eia.doe.gov/dnav/pet/pet_cons_psup_dc_nus_mbbl_a.htm), using data for May through August and half of September.

densely populated, however, generally do not experience large reductions in ozone pollution following the imposition of CARB. Fortunately, these counties are those for which ozone levels are typically below unhealthy levels.[38]

Using our estimated ozone reductions for California, a "back of the envelope" calculation indicates that the benefits of CARB gasoline significantly outweigh its costs. Using Bell et al.'s (2004) estimates of ozone's mortality impacts, our RD estimates translate into 660 saved lives in California each year.[39] Further, given California's 2008 gasoline consumption of about 14.8 billion gallons each year and estimates that CARB increases gasoline's production cost by 8-11 cents per gallon, CARB imposes a cost of approximately $1.2-$1.6 billion per year. These figures translate into a cost of $1.8-$2.4 million per life saved. This range is significantly lower than the EPA's official value of a statistical life of $6.45 million (2005 US$). If we were to factor ozone's morbidity and environmental impacts into this calculation, CARB gasoline's benefit to cost ratio would be pushed even further upwards.

If carried out on a county-by-county basis, our simple cost-benefit analysis would of course suggest that CARB should be required in only a select group of counties. However, a comprehensive policy analysis of spatial gasoline regulation must recognize the possibility that fine-tuned regulatory targeting may further segment the gasoline market, potentially increasing gasoline price levels and volatility. Further, CARB (and RFG) may convey health benefits through reductions in emissions of toxic air pollutants. These benefits are not evaluated in this paper and may be significant even in areas for which we do not observe substantial decreases in

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[38] These heterogeneous effects were estimated using changes in ozone concentrations at the time of CARB's introduction: March 1996. Our estimates may not accurately reflect CARB's *current* impact on ozone to the extent that ozone precursor emissions have changed since 1996.

[39] We calculate this impact by first averaging our monitor-level results to the county-level and assuming no effect in non-monitored counties. We then use Bell et al.'s (2004) formula, in combination with data on population and pre-CARB ozone levels, to compute county-level estimates of lives saved.

ozone levels. A full assessment the trade-offs involved in optimizing regional gasoline regulations is beyond the scope of this paper; however, the air quality benefits that we estimate here should serve as useful inputs to future research in this direction.

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