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Marginal Reductions in Vehicle Emissions Under a Dual-Blend Ethanol Mandate: Evidence from a Natural Experiment

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Abstract

Among the many reasons policy makers across the world have sought to supplement fuel supplies with ethanol-blended fuels are the cited environmental benefits that come with replacing a fossil-fuel with a cleaner burning alternative. Dual-blend ethanol mandates, in which multiple ethanol blends are simultaneously available, are one way policy markers can move forward with more aggressive mandates more quickly. The recent ethanol mandate in the state of New South Wales, Australia offers a unique natural experiment to quantify the potential environmental benefits and costs of a dual blend ethanol policy. This paper estimates the impact on carbon dioxide (CO_2) emissions from road-activity that are attributable to the implementation of the New South Wales ethanol requirements. We find that there was a decrease in emissions due to the policy, but that the decrease is relatively minor given the size of the market and that it comes at a high cost. The cost was over \$1,200 per ton of carbon to reduce gasoline emissions by just 1.2%.

JEL Classification Codes: L51, Q41, Q42, Q51

Keywords: carbon emissions, ethanol mandates, unintended consequences, cost-benefit

1 Introduction

Policy makers are faced with a variety of goals when confronting the multi-headed Hydra of domestic fuel policy. It is no surprise, then, that many countries have adopted ethanol mandates to confront the pressing issues of energy independence, volatile fuel costs, support of domestic industries and agriculture, and importantly – environmental concerns. Despite the recent fervor surrounding ethanol-use, though, few authors have discussed the consequences of heightened ethanol consumption. Heal (2010) comments on this and connects the lack of scholarship with the abundance of

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countries that promote the use of renewable energy sources before fully analyzing the potential for fulfilling policy objectives and the costs of doing so.

Given the multifaceted nature of ethanol policy adoption, a focused analysis of each piece of the ethanol policy puzzle is necessary. To aid policy makers when considering the adoption or reconsideration of an ethanol mandate, this paper seeks specifically to comment on the environmental benefit of an ethanol requirement in terms of carbon dioxide (CO_2) reductions from road activity, relative to the costs of achieving that benefit. We do so using observed market responses in a natural policy experiment setting.

We examine the emissions reductions induced by the ethanol mandate in New South Wales, Australia. Beginning in October of 2007 the state of New South Wales, Australia, required that a target proportion of total gasoline volumes include ethanol fuel. The first iteration of the mandate required 2% of the fuel supply be comprised of ethanol, and successive mandates increased the requirement to 4% and 6% of the total fuel supply.

The NSW mandate was relatively aggressive in the sense that regulators wanted to introduce ethanol blends while there was debate about its safety in about 20% of vehicles on the road. In order to move the mandate forward, regulators adopted a dual blend mandate – i.e. both E10 and a more expensive version of E0 were to remain simultaneously available. In this case, premium fuel would serve as the ethanol-free E0. The design was intended to move consumers whose vehicles could handle E10 onto E10, while ethanol-free fuel would remain available, albeit more expensively, only for those vehicles that needed it. The goal of the mandate was to replace all unblended regular gasoline with E10 within five years, while leaving premium fuel unaffected.

This is different from the adoption of E10 in the U.S. where E10 was implemented almost universally and E0 largely disappeared as a choice. It is more similar to the current (slower) transition from E10 to E15 in the U.S. made necessary by increases in the U.S. ethanol targets. As E15 is not suitable for all vehicles, both must remain available, and the cost of a RIN (Renewable Identification Number), paid by producers and blenders, creates a price wedge between E10 and E15 designed to move consumers on to E15 wherever possible. The EPA and automobile manufacturers disagree substantially on the percentage of vehicles that can safely use E15, however. The EPA has certified E15 for all 2001 and later model vehicles (about 70% of the fleet) while automobile manufacturers report that E15 is suitable only in some post-2012 models (about 10%) and that its use may damage engines and void manufacturer warranties. Consumer enthusiasm for E15 has been low and adoption has been poor. Similarly, E85 adoption by flex fuel vehicle owners has also been abysmal to date.

Unlike many other mandates national in scale, the Australian experience lends itself well to a controlled treatment effect analysis. The NSW mandate was not adopted nationwide, so other states within Australia that sell ethanol-blended fuels can serve as control groups, subject to the same federal policies and global influences, but absent a large scale ethanol mandate. We are thus able to distinguish between the change in emissions directly due to the passing of the ethanol mandate from what would have occurred naturally given existing market forces or relative price fluctuations. From this policy treatment effect analysis, we are able to quantify the amount of emissions that have been abated, or more accurately – avoided, due to the mandate-induced rise in ethanol-blended fuel consumption.

We seek to answer four questions: By how much did CO_2 emissions change in NSW as a direct result of the ethanol mandate? How much was the cost per ton of CO_2 abated? How does this cost compare to original expectations and to costs for comparable reductions in other parts of the world where a universal, rather than a dual-blend, mandate has been used? How does it compare to the cost of alternate methods of carbon reduction?

To preview results, we find that there is a statistically significant but globally small decrease in the amount of vehicle-emissions due to the policy, 14,501 tons of CO_2 per month. The reduction was marginal and well below the carbon reduction goals the state was hoping to achieve. The reduction amounted to 1.2% of monthly Australian emissions or about 0.0032% of average monthly CO_2 emissions from U.S. energy sources (in 2014). Yet for consumers, the small impact came at a sizable cost. Combining our results with information on the costs of implementation and production from government sources and information on increased pump prices from Noel and Roach (forthcoming), we find that consumers, taxpayers and firms ultimately paid between $1,276.50 and $1,407.53 per ton of CO_2 to make only a dent on emissions from gasoline; an expensive way to achieve a marginal improvement in environmental quality. This figure is substantially higher than that initially expected, higher than comparable international figures, and higher than that attainable from other means of attaining emissions reductions goals.

In particular, we contrast our figures to Meng et al. (2013). Meng simulates carbon reductions from a carbon tax and finds that a \$23 per ton carbon tax in Australia should should cause CO_2 emissions to decrease by up to 12%. In our paper, we find that the implementation of the dual blend ethanol mandate resulted in just 1/10th as much emissions reductions as that projected by Meng, and did so at a cost 56 to 61 times greater. The \$23 price tag used by Meng derives from the Australian Clean Air Regulator's regulated permit price for releasing a ton of carbon into the atmosphere at \$23 in 2012/2013. We conclude that the NSW ethanol mandate was exceptionally expensive relative to other methods of carbon reduction.

We proceed as follows. Section 2 gives additional background and insights from the recent literature on the costs of ethanol mandates. Section 3 discusses the data and the methodology used in the analysis. Section 4 presents results and Section 5 concludes.

2 Literature and Background

Ethanol mandates have become popular over the past decade and over sixty nations have implemented some form of one (GRFA (2014)). Purported benefits include environmental benefits, greater energy independence, and benefits to a domestic ethanol industry. The fact that ethanol production and consumption can affect a variety of industries is one of the reasons for its recent surge in popularity among policy makers (Charles et al. 2007).

However, mandates also come at a cost, a cost that is higher to the extent there are unintended consequences (Jaeger and Egelkraut 2011). For example, the passing of an ethanol mandate can potentially lead to higher land and food prices, increased use of water to grow the fuel source, and increased energy-use from polluting sources to process plant materials into fuel (Pimentel 2003; Jaeger and Egelkraut 2011; Carter et al. 2013; Wu and Langpap 2014). Griffin (2013) states that it is time to reconsider ethanol mandates in the U.S. because the realized benefits have been minimal. Griffin (2013) and Carter et al. (2013) further note that there are negative spillover effects from the U.S. ethanol mandate in developing countries in the form of higher food prices. Along the same lines, Drabik and De Gorter (2013) show that there is "leakage effect" and that emissions increase elsewhere in the world from U.S. fuel standard policies because oil prices decrease. Grafton et al. (2012) recognize this shift in equilibrium prices and induced demand for fossil-fuels as a green paradox. Charles et al. (2007) mention several potential drawbacks of developed countries championing a biofuels policy, including environmental drawbacks, and find the justification used by these governments to be "questionable."

Only a few other studies have measured the amount of emissions that have been mitigated due to ethanol consumption, though none from a treatment effect point of view. Szklo et al. (2005) find that 5.4 metric tons of CO_2 emissions per year are avoided in Brazil due to ethanol consumption. Nguyen and Gheewala (2008) find that in Thailand the consumption of biofuels leads to a 4.3% life-cycle decrease in emissions compared to gasoline. Greaker et al. (2014) use a numerical simulation to show that the introduction of a renewable fuel standard slows the rate of oil depletion which, in turn, causes more emissions. The authors also show that combining fuel standards with a biofuels subsidy leads to increased emissions. Grafton et al. (2012) finds that the opposite holds. Namely, that the adoption of new policies hurries the depletion rate of fossil fuels and causes more damage. Given the very long half-life of CO_2 emissions, though, this discrepancy in the delay in emissions is less meaningful in the scope of greenhouse gas accumulation over time. Even the Economist has contributed to the discussion on biofuels and their environmental impact by noting that the "biofuels that can best compete commercially are not, in fact, green," and that "those that are green cannot compete commercially." (Economist, 2015).

Paying attention to the relative costs of passing an ethanol policy Henke et al. (2005) show that the abatement costs of using ethanol in Germany are about ten times the cost of simply purchasing permits on the open market. The authors also note that "with the same economic effort a larger amount of [greenhouse gas] emissions could be avoided elsewhere"(Henke et al. 2005). Jaeger and Egelkraut (2011) find that the use of biofuels in the U.S. has had a negligible impact on greenhouse gas emissions, and that a gasoline tax would be more effective at reducing CO_2 emissions. Additionally, Holland et al. (2009) simulate the effects of low carbon fuel standards for the U.S. and find that fuel content mandates, like the Australian ethanol requirement, are a very expensive way to reduce emissions.

The present paper reaches a similar conclusion for the New South Wales mandate. We find that the ethanol mandate in New South Wales led to a small decrease in emissions relative to the total amount of emissions produced by gasoline, and does so expensively. We discuss how the dual blend nature of the New South Wales mandate can magnify the expense, resulting in an effective price per ton of carbon abated among the most expensive found in the literature. Against the backdrop of previous studies, the results of this paper show that, even within the universe of ethanol mandates, the manner in which they are implemented is important for the ultimate costs of the program.

In our analysis we abstract away from disagreements surrounding energy intensity issues in converting plant materials into a fuel source, and instead focus on the amount of emissions effectively taken out of the environment during road-use by consumers moving to an ethanol blended fuel. Yusef et al. (2011) mention that ethanol in Australia is carbon-neutral in combustion due to the fact that leftover fuel materials can be returned to the soil (Topgul et al. 2007). The authors do not discuss, however, the possibility that carbon-intensive fuel sources are used during the production process; e.g. using coal-generated electricity to process plant materials into a combustible fuel. Macedo et al. (2008) discuss the energy-intensive nature of Brazilian ethanol production. Pimentel (2003) and Patzek et al. (2005) find the net impact of ethanol from corn in the United States to be negative. Moreover, they find that other greenhouse gases are emitted alongside CO_2. In this light, the results of this paper can be thought of as an upper bound on emissions reductions following the ethanol mandate. If ethanol production is energy-intensive, and the energy source used when converting the plant material into fuel is carbon-intensive, then the net change in CO_2 would be less than the reduction calculated here and the cost higher (infinite if the net change is positive).

The only study, to our knowledge, to examine the New South Wales ethanol mandate was Noel and Roach (forthcoming). Noel and Roach focus on consumers with E10-incompatible (or believed to be E10-incompatible) vehicles and show that large numbers of consumers switched away from the new E10 blend to the more expensive, but ethanol-free, premium grade gasoline. They estimate the higher cost to these consumers specifically in terms of higher expenditures on fuel for the same amount of energy, but not other costs. Our focus instead is the mandate's effect on CO_2 emissions levels and on the cost of abatement not only in terms of cost to consumers, but also to producers and taxpayers. We also differ from the previous study methodologically in that we embed a difference-in-differences model inside a structural demand model to estimate carbon emissions reductions. We calculate supply and demand elasticities for gasoline as well and estimate the rate of passthrough of producer costs into consumer prices, which is relevant to avoid doublecounting of certain costs in the overall cost calculation.

The NSW mandate had four phases, in which ethanol was required to make up 2%, 4% and then 6% of the total fuel supply (corresponding to an E10 market share of 20%, 40% and 60%) and then finally replace regular unblended fuel at all pumps altogether. The final phase to remove all regular unblended gasoline was abandoned before it came into effect. To allow for the approximately 20% of vehicles for whom E10 was not recommended by manufacturers, according to a published list cited by the NSW government, premium grade fuel was to remain ethanol-free, i.e. E0, as a substitute.

Because ethanol-blended fuel and conventional gasoline, both regular and premium, are close physical substitutes to one another, one potential reaction to the mandate is that consumers will switch to still-available non-ethanol substitutes, thereby diminishing the effectiveness of the mandate. Indeed, E10 was never able to enjoy a high enough market share to fully meet the mandate requirements, and plateaued before reaching a 40% share of the market. Noel and Roach (forthcoming) discuss the wide-spread avoidance of E10 in New South Wales. Salvo and Huse (2013) uncover a similar phenomenon in Brazil in which many consumers did not switch to the readily available ethanol-blended fuel. Even during the transition to a universal E10 mandate in the U.S. and Canada, websites cropped up identifying and directing consumers to the shrinking number of stations still selling ethanol free gasoline.[1] Today, adoption of higher ethanol blends such as E15 and E85 in the U.S. continues to be abysmal, in spite of RIN-based cross-subsidization from lower blends such as E10 to higher blends such as E15 and E85 (Knittel et al. (2015)). In a similar way, the diversion from E10 fuel in NSW to a more expensive ethanol-free version added to the cost of the NSW mandate while, ironically, doing little for the goal of carbon dioxide emissions reductions since the alternative continued to be ethanol free.

[1] Ethanol-free gasoline, bottled and sold by the quart for two-cycle engine use sold for between \$5 and \$8 a quart in the U.S. in 2012 (\$19 and \$30 a gallon) when E10 sold for \$3.60 a gallon.

3 Data and Methodology

We employ data on gasoline volumes, prices, and other ancillary information on a state and monthly basis, before and after the start of the New South Wales ethanol mandate. Gasoline volume data is provided by the Australian Bureau of Resource and Energy Economics (BREE), price data is provided by Fueltrac and Informed Sources, wholesale price data (the "terminal gate price") is provided by Orima Research, and data on new vehicle registrations and the unemployment rate is provided by the Australian Bureau of Statistics. CO_2 emissions are reported in thousands of metric tons, new vehicle registrations are reported in thousands of new registrations, and the price of E10 and the terminal gate price are in dollars per liter. The monthly data spans from July 2005 to July 2013.

We are interested in the impact of the NSW mandate on carbon dioxide emissions, CO_2, and later, on the cost per ton of carbon dioxide emissions abated. Absent direct measurements of carbon dioxide emissions from vehicle use, the standard approach is to employ a conversion algorithm to calculate emissions from fuel use. The amount of carbon dioxide (CO_2) emitted in state s at time t through sales of grade g fuel is calculated as:

$$CO_{2,gst} = V_{gst} * (MJ/L)_{gst} * (CO_2/MJ)_{gst} \tag{1}$$

where V_{gst} is the volume of grade g fuel sold in state s at time t, in liters, $(MJ/L)_{gst}$ is the energy content factor of grade g fuel, measured in mega joules per liter, and $(CO_2/MJ)_{gst}$ is the emissions factor of the fuel, measured in metric tons per mega joule. Measures of the carbon factor and energy density are provided by the Australian Bureau of Infrastructure, Transport and Regional Economics (BITRE)[2]. Summing across the three primary grades of fuel, $g = \{$RULP, PULP, E10$\}$, the total direct carbon dioxide emissions from combusting fuel of all grades in state s at time t is calculated as:

$$CO_{2,st} = \sum_{g=1}^{3} CO_{2,gst} \tag{2}$$

[2]Carbon factors calculated by the U.S. Environmental Protection Agency (EPA) are similar. Our results are robust to this choice, emission estimates are not significantly different from one another. We report results using the BITRE calculation throughout.

We report basic summary statistics in Table 1.

table 1 goes about here

The analysis proceeds in two parts. In the first part, we estimate the effect of the mandate on CO_2 emissions in NSW. One simple approach would be to estimate the change in emissions in NSW before and after the introduction of the mandate. However, since CO_2 emissions can change independently of the mandate due to changes in, for example, the number and composition of vehicles on the road and general improvements in fuel efficiency over time, such an estimate would be infected by any such contemporaneous changes or trends stemming from other reasons.

Fortunately, the setting and data lends itself well to a difference-in-differences approach. NSW was the only state to implement an ethanol mandate. The other mainland states – Victoria, Queensland, South Australia, and Western Australia – had no such ethanol mandate, saw little in the way of E10 popularity, and serve as comparable control states, subject to the same federal policies and global trends as NSW, but absent a mandate. The differencing technique essentially compares the change in CO_2 emissions in NSW after the introduction of the mandate to the change in CO_2 emissions in the other states over the same period, with the effect of interest being the difference in the two changes. In this way, we "difference out" any unobserved shocks or trends common across states and estimate the effect of the mandate itself on NSW carbon emissions.

We model CO_2 emissions from road activity stemming from the use of a particular grade of gasoline, $CO_{2,gst}$, as:

$$\begin{aligned} CO_{2,gst} &= \beta_{0g} + \beta_{1g}NSW_s + \beta_{2g}STAGE1_t + \beta_{3g}STAGE2_t + \beta_{4g}STAGE3_t \\ &\quad +\beta_{5g}NSW_s * STAGE1_t + \beta_{6g}NSW_s * STAGE2_t + \beta_{7g}NSW_s * STAGE3_t \\ &\quad +\beta_{8g}PRICE_{gst} + X_{gst}B_g + \varepsilon_{gst} \end{aligned} \tag{3}$$

where NSW is a dichotomous variable equal to one when $s = NSW$ (New South Wales); $STAGE1$, $STAGE2$ and $STAGE3$ are dichotomous variables each equal to one from October 2007, January 2010, and October 2011, respectively, corresponding to the start of the 2%, 4% and 6% stages of the mandate. $PRICE_{st}$ is the price of fuel of grade g and the term X_{gst} includes demand-side determinants, in particular: contemporaneous and lagged values of the unemployment rate to account for changes in income, contemporaneous and lagged values of new vehicle purchases to control for the age of the vehicle fleet and the stock of vehicles, and monthly fixed effects.

The treatment effects for each of the three stages and the primary coefficients of interest are β_5, β_6 and β_7. The overall effect of the mandate in the 4% and 6% periods are the sums of the relevant coefficients: $\beta_5 + \beta_6$ and $\beta_5 + \beta_6 + \beta_7$, respectively.

As unobserved changes in demand for driving can affect both carbon emissions and the price of gasoline, prices are potentially endogenous. Therefore we estimate a set of first stage price equations (the "simple supply model") using wholesale prices as the instrument:

$$\begin{aligned} PRICE_{gst} &= Z_{gst}\Gamma_g + v_{gst} \\ &= \gamma_{0g} + \gamma_{1g}NSW + \gamma_{2g}STAGE1_t + \gamma_{3g}STAGE2_t + \gamma_{4g}STAGE3_t \\ &\quad + \gamma_{5g}NSW_s * STAGE1_t + \gamma_{6g}NSW_s * STAGE2_t + \gamma_{7g}NSW_s * STAGE3_t \\ &\quad + \varphi_g TGP_{gst} + \Pi_g + v_{gst} \end{aligned} \tag{4}$$

where $PRICE_{gst}$ and TGP_{gst} are the retail and wholesale prices respectively of grade g in state s at time, Π_g are monthly indicator variables, and v_{gst} is the stochastic error term.

Unlike retail prices, TGP is reasonably modeled as exogenous. Australia is a net importer of refined product (mainly from Singapore) and the cost of importing refined product determines the wholesale price on the margin. In fact, wholesalers set TGP according to a formula known as import pricing parity (IPP), essentially the price of refined product from Singapore (e.g. the mean Platts quote for Mogas 95 from Singapore), plus transportation costs, taxes, and other fixed adjustments. TGP and Asian wholesale prices are very highly correlated.

It is well known that there can be lags and asymmetry in the transmission of wholesale prices into retail prices (Borenstein, Cameron & Gilbert (1997), Noel (2009), Lewis (2009), Tappata (2009), Lewis & Noel (2011), and very many others). While such lags typically tend to be on the scale of a few days or weeks, we estimate a second price equation that allows for asymmetric and lagged responses to wholesale price changes. The Engle and Granger (1987) style vector autoregressive error correction model (VAR-ECM) is given by:

$$\begin{aligned}\Delta PRICE_{gst} &= \delta_{g,0} + \sum_{i=0}^{I} \delta^{+}_{g,1+i} \Delta TGP^{+}_{gs,t-i} + \sum_{i=0}^{I} \delta^{-}_{g,1+i} \Delta TGP^{-}_{gs,t-1} \\ &\quad + \sum_{j=1}^{J} \delta^{+}_{g,1+I+i} \Delta PRICE^{+}_{gs,t-j} + \sum_{j=1}^{J} \delta^{-}_{g,1+I+i} \Delta PRICE^{-}_{gs,t-j} \\ &\quad + \phi_g (PRICE_{gs,t-1} - Z_{gs,t-1}\Gamma_g) + X_{gst} B_g + v_{gst} \qquad (5)\end{aligned}$$

where $\Delta TGP^{+}_{gs,t-i} = \max(0, \Delta TGP_{gs,t-i})$, $\Delta TGP^{-}_{gs,t-i} = \min(0, \Delta TGP_{gs,t-i})$, and $\Delta PRICE^{+}_{gs,t-j}$ and $\Delta PRICE^{-}_{gs,t-j}$ are similarly defined. The error correction term, in parentheses on the last line, measures the long run, steady state relationship in levels between retail price and terminal gate prices. Z_{gst} was earlier defined in Equation 4 and includes treatment, period, and treatment-period interactions, as well as TGP and monthly dummies. We add $PRICE_{gs,t-1}$ to both sides to obtain the final first stage set of equations in the VAR pricing model and use the predicted values of these models for $PRICE$ in our main estimation.

We compare the observed and counterfactual worlds to derive an estimate of the amount of emissions reductions specifically due to the NSW mandate. Specifically, we calculate the amount of emissions from observed E10 volumes under the successive mandates, and compare this to the counterfactual amount of emissions that would have been realized in the absence the mandate i.e. if E10 had the same emissions factor as conventional gasoline.

Lee (2005) shows that the estimated treatment effect would not be valid if the treatment group (NSW) and the control group do not share a "common trend" for the variable under study. For example if E10 consumption (and emissions) were growing (declining) at a different pace than the control group prior to the ethanol mandate, for some other reason, then the estimated treatment effect would be biased. We evaluate the common trend assumption and fail to reject the hypothesis that NSW had a different trend than the control states prior to the mandate, both in terms of volumes and in terms of prices. We also conduct a battery of panel-unit root tests and conclude that the time series are stationary both before the mandate and within each stage.

The second part of the analysis relates the total amount of emissions abated to a set of costs associated with abating them, and compares the per ton cost to other benchmarks. We examine three, potentially overlapping, costs. First are the costs to consumers in terms of the higher expenditures required to purchase the same amount of energy contained in the fuel as before. Next are the fixed costs to producers from implementation and retrofitting, and third are the costs to taxpayers in support of ethanol production subsidies. We do not suggest that there are no other costs, but these three make up the largest, most cited, and easiest costs to quantify.

The first set of cost estimates comes from Noel & Roach (forthcoming), the remaining two derive from government sources. We note that these costs can be overlapping – e.g. costs to producers or subsidies paid by taxpayers can be passed through to prices paid by consumers. We use elasticity and passthrough estimates to measure the incidence and passthrough rates of costs into final prices, and adjust our total cost estimates as necessary to avoid doublecounting. The elasticities we estimate are potentially of independent interest in and of themselves.

We compare the costs per ton of CO_2 abated under the mandate to both the cost of CO_2 emission reductions by other means, and the cost of emissions reductions of a similar nature in other parts of the world.

4 Results

Tables 2A, 2B, and 2C show the estimated impact of the NSW ethanol policy on carbon emissions, for regular, E10, and premium, respectively. Specification (1) reports the second stage carbon equation without additional controls and Specification (2) reports it with additional demand controls. Specification (3) reports the carbon equation with additional demand controls but uses the VAR-ECM model in place of the simple supply model in the first stage. Specifications (4), (5), and (6) report results for E10 and Specifications (7), (8), and (9) report results for premium grade gasoline.

tables 2A-2C go about here

Results across the three models are similar. Consider Specifications (2), (5), and (8), which include additional demand controls and use the first stage simple supply model to instrument for price. In Specification (2), we find carbon emissions from the sale of unblended regular gasoline fell by 96,729 tons per month on average in the 2% mandate period, an additional 302,598 tons in the 4% mandate period, and an additional 205,091 tons per month in the 6% mandate period. As consumers switched from unblended regular to E10, carbon emissions from the sale of E10 increased by 111,989 tons per month in the 2% period, an additional 204,047 tons in the 4% period, and an additional 53,765 tons in the 6% period (Specification (4)). As Noel and Roach (forthcoming) point out, one of the consequences of the NSW dual blend mandate was that a large proportion of consumers switched to the still ethanol-free premium grade gasoline instead of to E10. From Specification (6), we see that carbon emissions from the sale of premium grade gasoline did not significantly change in the 2% period, but then increased by 110,374 tons per month in the 4% period and by an additional 70,964 tons in the 6% period. The mandate resulted in a substantial change in the composition of fuels.

table 3 about here

Table 3 reports results from the first stage price regressions, both for the simple supply model and the VAR-ECM model. Specifications (10) and (11) report results for regular gasoline, Specifications (12) and (13) for E10, and Specifications (14) and (15) for premium grade gasoline. The table shows that wholesale prices (TGP) are the primary driver of retail prices.[3] The t-statistics range from 45 to 247 and the F-statistics of the first stage regressions are extremely high. The specifications also show that the mandate itself had no effect on the prices of any of the three grades – all interaction coefficients for the 2%, 4%, and 6% periods are statistically insignificant.

There are two ways in general that an ethanol mandate can create a reduction in carbon emissions. The first is by changing the composition of fuels sold from lower (or zero) ethanol blends to higher ethanol blends, like E10, while holding total energy-adjusted gasoline volumes constant. The second, to the extent that the mandate causes fuel to be more expensive generally or forces consumers to switch to more expensive fuels, is by lowering the aggregate volume of gasoline sold. In other words, even if E10 had the same carbon content as unblended fuel, the mandate could reduce emissions if it causes causes consumers to drive less, perhaps due to higher prices.

[3]For space considerations, we report the full set of VAR-ECM coefficients, including lagged price, lagged cost, and the series of lagged price and cost changes, in Appendix table A1.

The results show that the mandate led to a statistically and economically significant change in the composition of fuels, away from unblended regular and towards E10 (and, collaterally, towards premium fuels as well). However, we find the mandate had no statistically significant effect on emissions through a reduction in the total (energy-adjusted) amount of gasoline sold. In other words, if we assume that E10 had the same carbon content as other fuels (ignoring the composition effect), we find there is no statistically significant reduction in emissions. The fact that consumers did not drive less after the introduction of the mandate is consistent with the very inelastic demand curves we estimate, below.[4] The only statistically significant reduction in emissions comes through the composition effect and therefore, we focus on this mechanism hereafter.

table 4 about here

We combine the grade-specific estimates to estimate the total reduction in carbon emissions for producing an equivalent amount of energy as would occur in the absence of the NSW mandate. Emission reductions come from the fact that E10 emits less carbon during road use than an energy-equivalent amount of unblended gasoline. Results are reported in the first column of Table 4. We find that each successive increase in the ethanol requirement led to a decrease in emissions relative to the baseline scenario in which conventional fuel was used.

Again, consider the demand model with additional controls combined with the first stage simple supply model. In the 2% policy period emissions fell by 4,403 tons per month due to the policy. In the 4% period emissions decreased by an additional 8,023 tons per month due to the policy, for a cumulative decrease of 12,426 tons of CO_2 per month. In the 6% period an additional 2,075 tons of CO_2 were reduced per month compared to the pre-mandate period, for a cumulative reduction of 14,501 tons of CO_2 per month as of the time of the 6% period. Each impact on the amount of emissions reductions from switching to E10 from conventional is statistically significant.

[4]The carbon savings when a consumer switches from a liter of unblended regular to nothing is greater than the carbon savings when a consumer switches from a liter of unblended regular to E10. We find a small reduction in volumes upon the introduction of the mandate (i.e. a small reduction in carbon even if E10 had the same carbon content as unblended gasoline). However, the estimate is statistically insignificant and the reduction through volumes cannot be causally attributed to the mandate. The point estimate yields a 38,870 ton reduction in carbon (compared to a standard deviation of 48,537 tons in NSW) due to the small volume decrease post-mandate.

We find similar results whether or not we include the additional demand controls and whether we use the simple supply model or the VAR-ECM model in the first stage equations. Using the VAR-ECM model in the first stage, and including the full set of demand controls in the second stage, we find a cumulative reduction of 14,406 tons of CO2 per month across all grades by the time of the 6% period. Without the demand side controls and using the simple supply model, we find a reduction of 14,516 tons.

Putting aside the question of carbon intensity in ethanol production, the results show that the ethanol mandate reduced the emissions emitted by vehicles as they drive. But can the impact of the NSW ethanol mandate be considered successful?

A cumulative decrease in emissions of 14,501 tons per month amounts to just a 1.2% reduction in the average emissions in New South Wales. Comparing this figure to the standard deviation of total CO_2 emissions for the entire sample, 339,304 tons, or the standard deviation of total CO_2 emissions for New South Wales alone, 48,537 tons, it is evident that the amount of CO_2 reductions directly due to using E10 instead of conventional gasoline is fairly small even in comparison to the normal fluctuations in CO_2 emissions emanating from New South Wales.

A gauge of the size of the impact on CO_2 emissions due to the NSW ethanol mandate can be seen in comparison with emissions levels in the United States. Because CO_2 is a trans-boundary pollutant, avoided emissions from Australia are welcome worldwide. In 2012, the amount of avoided emissions in NSW is equivalent to a 0.01% reduction in CO_2 from motor-gasoline in the U.S., or a 0.0032% reduction in total U.S. CO_2 emissions from energy sources.

New South Wales is, of course, smaller than the United States. The population of NSW, Australia, is 7.5 million (2013 figure), or about 1/42 of the 316.5 million (2013 figure) population of the U.S. However, if we scale up the population of NSW to be equivalent to that of the U.S. and scale up the emissions reductions from the NSW mandate accordingly, the amount of avoided emissions from the NSW mandate would still only be equivalent to a 0.135% (one-seventh of one percent) reduction in U.S. emissions.

Figure 1, below, shows the time-series of monthly emissions reductions, or avoided emissions, in NSW due to those that switched to E10. The emissions time series indicates that emissions have fallen due to E10 substituting for conventional fuel as expected. Following the 2% ethanol mandate emissions gradually decline until midway through the 4% mandate period. The 6% mandate that went into effect in October of 2011 had no effect on the consumption of E10 and hence the emissions reductions leveled off at about 15,000 tons of CO_2 per month.

figure 1 about here

If the above emissions reductions in New South Wales came at no cost, and the emissions reductions found here held up after accounting for life-cycle issues in ethanol production, the policy could reasonably be called successful in reducing CO_2 regardless of the magnitude. That is to say, if emissions reductions were a free by-product of transferring consumers to ethanol-blended fuels, then any ethanol policy is a Pareto improvement on current conditions. However, the total reductions are modest and yet the policy does come with several important costs that must be considered in the overall analysis.

First, there was a taxpayer-funded ethanol production credit of 38.143 cents per liter of pure ethanol, yielding a taxpayer cost of 3.8143 cents for each liter of E10 sold, or about $6.91 million a month for E10 sold in NSW as of 2013. Given a 14,501 ton monthly reduction in CO_2 emissions, that translates to a cost of $476.52 per ton of CO_2 abated. Further, Noel and Roach (forthcoming) found that the mandate led to higher fuel costs to consumers in part because the energy-adjusted price of E10 was higher, even with the subsidy, and in part because many consumers had to, or chose to, avoid the E10 blend altogether to buy more expensive but ethanol-free alternatives. The cost was $11.6 million per month by 2013, or $139.2 million for the year. This is equivalent to spending an additional $799.94 per ton of CO_2 mitigated. The sum of these two components is $1,276.50 per ton.

Next are the costs to firms. There are widely varying estimates of the cost of retrofitting stations, pumps and tanks for E10. They range from a few thousand dollars for conversion of newer, well maintained tanks to up to $800,000 per station for full tank replacement of older tanks (excluding lost profits during the retrofit). Industry estimates for the total changeover cost have been estimated to be between $124 million and $270 million dollars (E10 Task Force, 2011). Assuming a risk-adjusted market interest rate of 8% and straight line depreciation over a 30 year useful lifetime for underground storage tanks, the carrying costs of the retrofit expense amounts to between \$14.05 million and \$30.6 million a year.[5] The median estimate is \$22.9 million a year, \$1.9 million a month, or \$131.03 per ton of CO_2 mitigated. The total of all three sources of costs amounts to \$1,407.53 per ton.

It is possible that retrofitting costs may be doublecounted in the calculation, both as a producer and a consumer cost, since consumer costs are measured as changes in overall gasoline expenditures. To the extent that retrofitting costs were already passed through to pump prices, those retrofitting costs should not be added again. It is also possible that, to the extent that the taxpayer funded subsidy was *not* passed through to consumers in the form of lower prices, the amount of the producer-retained subsidy should be deducted from producers' costs to get a net producer cost. Inversely, to the extent the subsidy *was* passed on to consumers in the form of lower prices (than otherwise would have occurred), no producer cost offset is warranted. The taxpayer cost remains an independent cost in any event.

The rate of passthrough of costs into final prices can be estimated in several ways. First, we consider a change in marginal costs, such as occurs with the ethanol production credit. One quick method to gauge passthrough in the industry is simply to examine the coefficient on wholesale prices (TGP) in Specifications (2), (4), and (6), above. The coefficients show close to complete passthrough of wholesale costs into retail prices month to month (97.2% on average). Results from the VAR pricing model, which relaxes the assumption of instant passthrough, shows that passthrough is in fact largely complete after two months.

A preferable and more structural approach to estimate marginal cost passthrough is to examine the relative size of supply and demand elasticities. Standard supply and demand equilibrium give rise to the following simple comparative statics identity:

$$Q^d(p(a)) = Q^s(p(a) - a) \tag{6}$$

where a is a shock to marginal costs, $p(a)$ is the price which depends on cost a, and $p(a) - a$ is the amount retained by the seller after it pays a in additional costs. The price $p(a)$ adjusts in the short run and more fully in the long run to ensure supply equals demand for any a. Totally differentiating

[5] The useful lifetime for underground storage tanks (UST) is typically 20 to 30 years depending on the type and date of installation.

with respect to a yields the following incidence equation:

$$\frac{dp}{da} = \frac{\partial Q^s/\partial p}{\partial Q^s/\partial p - \partial Q^d/\partial p} = \frac{\eta}{\eta - \varepsilon} \tag{7}$$

where η is the price elasticity of supply and ε is the price elasticity of demand. Essentially, when the price elasticity of supply is high relative to the price elasticity of demand (in absolute value), dp/da tends to one and there is complete passthrough. When the price elasticity of demand is high relative to the price elasticity of supply, dp/da tends to zero and there is no passthrough.

We modify Equations 3 and 4 to estimate a simple structural supply and demand model from which we estimate price elasticities. In particular, we add total volume as an explanatory variable in Equation 4 and replace CO_2 with total volume on the left hand side of Equation 3.

We find the price elasticity of demand implied by the model estimates are -0.15, -1.84, and -0.54, for regular, E10, and premium, respectively. We find the price elasticity of supply is equal to 53.4, 70.9, and 43.6, respectively. Not surprisingly, the price elasticity of supply in the industry is highly elastic and much larger than the price elasticity of demand (in absolute value). This yields an average passthrough estimate of $dp/da = 0.993$ for marginal cost increases into gasoline prices, insignificantly different from one (i.e. complete passthrough).

We conclude that changes in marginal costs, such as the ethanol production credit, tend to be passed through almost fully to consumers. In other words, no producer cost offset is necessary, consumer benefits from the subsidy are likely already incorporated in the estimate of consumer cost, and the taxpayer cost of the subsidy program remains as an independent cost. Even if we were to conservatively assume that only 95% of the subsidy were passed through, it would reduce our overall estimates by only $23.83 per ton, and none of our conclusions would change.

Retrofitting costs are different insofar as they are not marginal costs but rather fixed costs. Passthrough of fixed costs is a longer term process. In perfectly competitive industries, changes in fixed costs are passed through only as firms exit and breakeven margins adjust higher. In imperfectly competitive industries, the cost may never be passed through. At a given time t, let λ_t be the rate at which fixed cost changes have been passed through to retail prices. Given the mandate is relatively new, we expect that λ_t is relatively close to zero – i.e. that retrofitting costs are unlikely to have been passed through to retail prices at present. The mandate includes an exemption for small retailers who would be most susceptible to losses and exit, and no unusual exit has been noted since the program began.

In the right two columns of Table 4, we report the combined costs per ton to all three groups for our baseline case of no passthrough $\lambda_0 = 0$ as well as for the case of complete passthrough, $\lambda_0 = 1$, at the present time $t = 0$. The overall cost is higher under $\lambda_0 = 0$ since retrofitting costs will not already be included in the consumer cost and thus represents an independent cost. Our general conclusions hold regardless of the value of λ_0. If $\lambda_0 = 0$, the total overall cost under our baseline model with controls is $1,407.53 per ton of carbon mitigated and if $\lambda_0 = 1$, the total cost is $1,276.50 per ton of carbon mitigated. Using the VAR-ECM model on the supply side instead of the simple supply model, we find the costs range from $1,284.89 per ton and $1,415.92 per ton. Using the simple supply model without additional demand controls yields $1,275.14 to $1,406.17. There is no significant difference between the models.

Our findings are in line with previous research that indicates that ethanol requirements are a very expensive way to reduce emissions (Henke et al. 2005; Jaeger and Egelkraut 2011; Holland et al. 2009). In fact, considering any one of the three costs individually – the taxpayer, consumer, and producer cost – the cost per ton of CO_2 abated is still very high ($476.52, $799.94 and $131.03 per ton of CO_2, respectively).

To put these figures in perspective, a comparison of the carbon cost under the NSW ethanol mandate to other Australian initiatives is useful. Australia is among the nations that signed the Kyoto protocol, and as such, they introduced legislation that put a price on CO_2 emissions in order to discourage carbon-intensive consumption. Australia's Clean Energy Regulator (CER) dictated that "liable entities" must purchase a non-bankable, non-transferable permit at a fixed price of $23 per ton of CO_2 released in 2012-2013 (CER 2014). At $\lambda_0 = 0$, our estimated cost of $1,407.53 per ton of CO_2 in 2013 exceeds this regulator-determined permit price for a ton of CO_2 emissions in 2013 by a factor of 61. The taxpayer cost alone exceeds it by a factor of 21, the producer cost alone exceeds it by a factor of 6, and the consumer cost alone exceeds it by a factor of 35. At $\lambda_0 = 1$, our estimated cost of $1,276.50 per ton exceeds it by a factor of 56.

Meng et al. (2013) simulated the effects of the Australian permit system and found that the

\$23 carbon permits should cause CO_2 emissions to decrease by up to 12%. In our paper, we find just 1/10th the emissions reductions due to the NSW ethanol mandate even though the cost was 56 to 61 times greater than the permit price.

We can alternately use the \$23 permit price to calculate the amount of money expected to be "saved" by the ethanol mandate in terms of the economic benefits from CO_2 reduction. Our estimates indicate that a total of 174,012 tons of CO_2 emissions were foregone in 2012 due to the policy. This amounts to a "savings" of \$4.0 million in 2012. Yet the cost of the mandate in 2012 was at least \$222 million conservatively including only taxpayer and consumers costs ($\lambda_0 = 1$). Compared to the permit price for a ton of CO_2, the costs of the mandate overwhelmingly outweigh the benefits in terms of foregone permit purchases. Further, imagining that motorists were the "liable entities" charged with purchasing permits for their CO_2 consumption, the per-vehicle savings of E10 use from the mandate in terms of forgone permit purchases was \$0.82 while the cost per-vehicle was \$45.59. We conclude that the costs of the mandate are exceptionally high.

To gauge the robustness of our main result and conclusions, we estimate a series of alternate models that estimate the amount of carbon reduction due to the mandate and its associated costs. We report the results in the bottom part of Table 4.

Our baseline models include an indicator variable equal to one for NSW and equal to zero for control states. In the first robustness check, we modify this to include a separate indicator variable for every state ("State Specific Indicators"). For this modification, we find a statistically significant reduction of 14,694 tons of carbon per month at a cost ranging from \$1,259.58 to \$1,390.71 per ton.

Second, we consider a pure reduced form difference-in-differences model similar to Equation 3 but without prices ("Reduced Form Model"). In this case, we find a statistically significant reduction of 14,540 tons of carbon per month at a cost ranging from \$1,273.06 to \$1,404.09 per ton.

Third, we address a general concern with difference-in-differences models pointed out by Bertrand, Duflo and Mullainathan (2004). Bertrand et al. show in Monte Carlo simulations that a simple OLS implementation of difference-in-differences can result in standard errors that are too low. This can lead to overrejection of the null hypothesis, especially when the time dimension is long and the number of clusters, or groups, is low. As our data is at the monthly level and the time dimension is relatively short, excessive serial correlation bias is not likely to be a problem. Bertrand et al.

find that much, though not all, of the potential bias can be resolved by clustering standard errors by groups or, in this case, states. In our paper, all reported results are clustered at the state level already.

To address any possible remaining tendency to overreject, the authors find the method that tends to work best with small numbers of clusters is to largely remove the time dimension by collapsing the data into a simple before period and after period (in our case, one before period and one period for each stage of the mandate). In the specification "Bertrand et al. Time Collapse", we estimate the model with the only 20 remaining datapoints per grade after collapsing the data. The specification is especially demanding (losing potentially good variation as well, and keeping only 4% of the original number of observations) so that the standard errors are necessarily higher. Yet we continue to find a statistically significant reduction of 14,516 tons of carbon, at better than the 1% level, and a cost of $1,275.15 to $1,406.18 per ton.

Along similar lines, we also estimated the model via bootstrapping with ten thousand repetitions ("Bootstrapped"). We find only a slightly higher standard error than in the baseline model, and the estimate of 14,501 tons of carbon avoided continues to be significant at better than the 1% level.

5 Conclusion

In summary, across the various models, we find the NSW ethanol mandate eliminated between 14,406 to 14,694 tons of CO_2 per month from road-users by the time of the 6% period. The reduction is relatively small yet came at a substantial cost. We estimate that the combined cost to consumers, taxpayers, and firms, across ranged between $1,259.06 and $1415.92 per ton of carbon avoided. These figures are 55 to 62 times higher than the $23 permit price set by the Australian Clean Energy Regulator in national efforts to reduce carbon emissions across other industries. We conclude that the dual blend NSW ethanol mandate has been an exceptionally expensive way to reduce carbon emissions. Policy makers are advised to keep this in mind when weighing environmental benefits alongside other policy goals that accompany the introduction of an ethanol mandate.

This point is made more obvious because the reduction estimate presented here is an upper bound of the reduction in CO_2 emissions. The processing of plant materials into usable fuel for vehicles is relatively energy intensive and our model abstracts from firm-level energy demand and emissions. Thus, what we have calculated is the amount of emissions reductions due to the ethanol mandate assuming that no additional CO_2 emissions were produced in the ethanol production process. In any scenario, the cost per ton of CO_2 avoided can come at a hefty price. We conclude that it is important for policy makers to understand the potential costs as well as benefits of an ethanol mandate as they weigh through the many choices available to them to try to meet the plethora of different goals they have for energy policy moving forward.

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Table 1. Summary Statistics

	Mean	Std. Dev.	Minimum	Maximum
Regular Emissions	513.24	248.22	190.67	1054.57
E10 Emissions	62.84	112.77	0.00	463.41
Premium Emissions	141.59	97.85	27.05	470.50
Regular Price	126.69	16.14	89.49	162.59
E10 Price	128.92	12.31	101.50	159.89
Premium Price	136.01	17.08	96.63	169.19
Terminal Gate Price	120.96	15.30	82.78	154.93
New Vehicle Registrations	16.23	8.11	4.13	37.19
Unemployment Rate	5.00	0.73	2.70	6.60

Emissions in thousand tons per month. Prices, including terminal gate prices, in Australian cents per liter (approximate exchange rate 1 AUD = 0.95 USD from 2009 to 2013.) New vehicle registrations in thousands of vehicles per month.

Table 2A. Mandate Impact on Emissions of Regular Grade Fuel

Dep. Var. = CO2	(1)	(2)	(3)
STAGE 1*NSW	-113.932*	-96.729*	-96.533**
	(25.366)	(24.787)	(14.282)
STAGE 2*NSW	-292.029**	-302.598**	-301.075**
	(18.234)	(19.593)	(17.470)
STAGE 3*NSW	-152.152**	-205.091**	-201.009**
	(12.451)	(7.961)	(18.104)
NSW	453.015*	76.365	76.073**
	(135.632)	(31.080)	(10.083)
STAGE 1	-43.470	-44.503	-44.081**
	(29.958)	(20.063)	(7.765)
STAGE 2	-37.381	-63.431*	-65.290**
	(17.640)	(18.944)	(9.217)
STAGE 3	21.687	9.027	11.234
	(12.539)	(8.939)	(9.663)
PRICE	-0.458	-0.648	-0.725**
	(0.525)	(0.292)	(0.245)
NEW VEHICLES		16.581**	15.298**
		(0.749)	(1.539)
LAGGED NEW VEHICLES		13.948**	15.143**
		(0.352)	(1.550)
UNEMPLOYMENT		22.594	18.396
		(30.256)	(23.123)
LAGGED UNEMPLOYMENT		-13.649	-7.743
		(25.681)	(22.880)
MONTHLY DUMMIES	Y	Y	Y
R-SQUARED	0.384	0.957	0.957
NUM. OBS.	575	570	545

Clustered robust standard errors in parentheses. * Significant at 5% level, **

Table 2B. Mandate Impact on Emissions of E10 Fuel

Dep. Var. = CO2	(4)	(5)	(6)
STAGE 1*NSW	110.917**	111.989**	98.714**
	(23.423)	(23.244)	(25.252)
STAGE 2*NSW	202.134**	204.047**	199.469**
	(8.510)	(8.674)	(18.710)
STAGE 3*NSW	55.644*	52.765*	67.573**
	(18.610)	(18.177)	(21.704)
NSW	2.167	-28.628	-21.576
	(7.156)	(38.894)	(23.560)
STAGE 1	32.532	27.233	17.174
	(25.585)	(23.841)	(15.422)
STAGE 2	10.647	9.012	18.306
	(8.233)	(8.812)	(11.002)
STAGE 3	-13.855	-19.439	-28.933*
	(15.943)	(19.390)	(12.469)
PRICE	-0.548	-0.336	0.010
	(0.233)	(0.228)	(0.415)
NEW VEHICLES		1.387	2.146
		(1.340)	(2.036)
LAGGED NEW VEHICLES		0.883	0.063
		(1.171)	(2.049)
UNEMPLOYMENT		34.856	99.552**
		(23.126)	(35.460)
LAGGED UNEMPLOYMENT		-28.999	-93.862**
		(21.678)	(34.122)
MONTHLY DUMMIES	Y	Y	Y
R-SQUARED	0.848	0.866	0.826
NUM. OBS.	485	485	268

Clustered robust standard errors in parentheses. * Significant at 5% level, **

Table 2C. Mandate Impact on Emissions of Premium Fuel

Dep. Var. = CO2	(7)	(8)	(9)
STAGE 1*NSW	-1.290	0.909	0.960
	(1.815)	(2.680)	(5.342)
STAGE 2*NSW	113.073**	110.374**	110.066**
	(2.523)	(3.048)	(6.010)
STAGE 3*NSW	83.795**	70.964**	71.881**
	(5.319)	(3.596)	(6.224)
NSW	116.815*	27.984*	26.823**
	(26.871)	(9.443)	(4.119)
STAGE 1	13.027**	11.775**	8.400**
	(1.756)	(2.512)	(2.874)
STAGE 2	7.201	2.505	3.146
	(2.898)	(3.341)	(3.199)
STAGE 3	20.970**	17.028**	13.112**
	(5.919)	(5.160)	(3.660)
PRICE	-0.465**	-0.498**	-0.287*
	(0.069)	(0.052)	(0.116)
NEW VEHICLES		4.251**	4.476**
		(0.488)	(0.553)
LAGGED NEW VEHICLES		3.128**	2.898**
		(0.541)	(0.557)
UNEMPLOYMENT		27.749	33.713**
		(19.112)	(8.470)
LAGGED UNEMPLOYMENT		-25.783	-30.430**
		(17.637)	(8.403)
MONTHLY DUMMIES	Y	Y	Y
R-SQUARED	0.752	0.969	0.967
NUM. OBS.	485	485	480

Clustered robust standard errors in parentheses. * Significant at 5% level, **

Table 3. First Stage Price Regressions

Dep. Var. = Price	Regular		E10		Premium	
	(10)	(11)	(12)	(13)	(14)	(15)
	Simple	VAR-ECM	Simple	VAR-ECM	Simple	VAR-ECM
STAGE 1*NSW	-0.513	0.095	-1.063	-0.153	-0.395	0.283
	(0.360)	(0.420)	(0.425)	(0.688)	(0.718)	(0.417)
STAGE 2*NSW	0.793	-0.034	0.356	-0.025	0.748	0.433
	(0.542)	(0.515)	(0.145)	(0.512)	(0.601)	(0.469)
STAGE 3*NSW	-0.786	-0.189	-0.933	-0.245	-0.995	-0.667
	(0.659)	(0.535)	(0.576)	(0.598)	(0.852)	(0.486)
NSW	0.903	0.095	-0.019	-0.141	1.873	0.117
	(0.764)	(0.269)	(1.117)	(0.614)	(0.760)	(0.299)
STAGE 1	2.233**	0.608**	3.432**	1.244**	5.172**	1.582**
	(0.358)	(0.232)	(0.460)	(0.409)	(0.691)	(0.234)
STAGE 2	0.732	0.069	0.470	0.323	1.378	-1.264**
	(0.547)	(0.324)	(0.219)	(0.418)	(0.607)	(0.315)
STAGE 3	1.086	0.131	0.996	0.218	3.199*	1.998**
	(0.656)	(0.298)	(0.665)	(0.370)	(0.849)	(0.300)
TGP (Wholesale)	0.985**	†	0.962**	†	0.933**	†
	(0.004)		(0.021)		(0.010)	
MONTHLY DUMMIES	Y	Y	Y	Y	Y	Y
R-SQUARED	0.983	0.990	0.974	0.986	0.977	0.988
NUM. OBS.	570	545	288	268	570	480

Clustered robust standard errors in parentheses. * Significant at 5% level, ** Significant at 1% level.

† Complete coefficients for the ECM-VAR model, including price and TGP lagged changes are reported in thet appendix.

Table 4. Emissions Reductions and Costs under Alternate Models

	Emissions reduction	Cost per ton ($\lambda_0 = 1$)	Cost per ton ($\lambda_0 = 0$)
Baseline Specifications			
With Simple Supply, No Additional Demand Controls	-14.516	$ 1,275.14	$ 1,406.17
	(0.518)	(43.94)	(48.45)
With Simple Supply, With Additional Demand Controls	-14.501	$ 1,276.50	$ 1,407.53
	(0.535)	(45.43)	(50.10)
With ECM-VAR Supply, With Additional Demand Controls	-14.406	$ 1,284.89	$ 1,415.92
	(0.467)	(40.35)	(44.44)
Alternate Specifications			
State Specific Indicators	-14.694	$ 1,259.68	$ 1,390.71
	(0.413)	(34.44)	(38.06)
Reduced Form Model	-14.540	$ 1,273.06	$ 1,404.09
	(0.501)	(42.38)	(46.75)
Bertrand et al. Time Collapse	-14.516	$ 1,275.15	$ 1,406.18
	(1.803)	(140.89)	(155.38)
Bootstrapped (10,000 reps)	-14.501	$ 1,276.50	$ 1,407.53
	(0.556)	(47.11)	(51.95)

Asterisks are suppressed as all results in the table are significant at better than the 1% level. Cumulative emissions reduction shown with clustered robust standard errors. Emissions reported in thousands of tons.

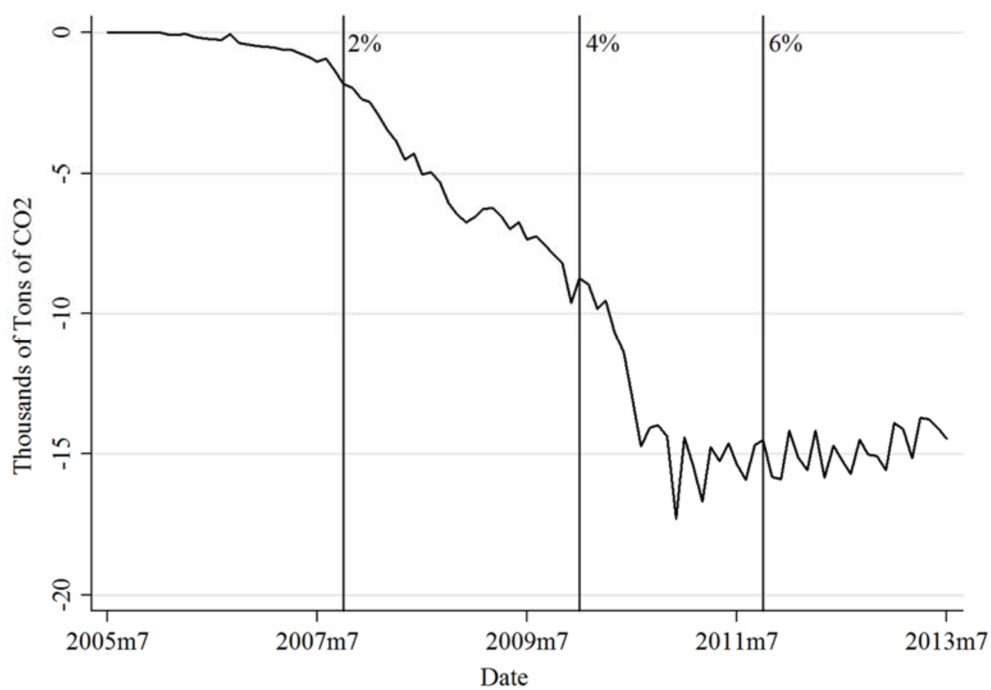

Figure 1. Monthly Emissions Reductions under the NSW Mandate

Table A1. Additional First Stage VAR-ECM Model Results

Dep. Var. = PRICE	Regular (10)	E10 (11)	Premium (12)
ΔTGP^{+}_{t}	0.920**	0.911**	0.580**
	(0.029)	(0.049)	(0.031)
ΔTGP^{+}_{t-1}	0.455**	0.471**	0.238**
	(0.066)	(0.094)	(0.055)
ΔTGP^{+}_{t-2}	0.448**	0.511**	0.267**
	(0.065)	(0.090)	(0.052)
ΔTGP^{+}_{t-3}	0.199**	0.223*	0.098*
	(0.063)	(0.087)	(0.049)
ΔTGP^{+}_{t-4}	0.111*	-0.058	0.017
	(0.056)	(0.082)	(0.044)
ΔTGP^{-}_{t}	0.904**	0.965**	0.731**
	(0.024)	(0.037)	(0.025)
ΔTGP^{-}_{t-1}	0.724**	0.657**	0.362**
	(0.069)	(0.101)	(0.060)
ΔTGP^{-}_{t-2}	0.265**	0.251*	0.264**
	(0.076)	(0.110)	(0.061)
ΔTGP^{-}_{t-3}	0.212**	0.140	0.314**
	(0.072)	(0.110)	(0.060)
ΔTGP^{-}_{t-4}	0.200**	0.130	0.221**
	(0.066)	(0.096)	(0.054)
$\Delta PRICE^{+}_{t-1}$	-0.428**	-0.305**	-0.031
	(0.061)	(0.083)	(0.066)
$\Delta PRICE^{+}_{t-2}$	-0.348**	-0.329**	-0.334**
	(0.062)	(0.082)	(0.067)
$\Delta PRICE^{+}_{t-3}$	-0.205**	-0.159*	-0.141*
	(0.060)	(0.080)	(0.066)
$\Delta PRICE^{+}_{t-4}$	-0.087	0.029	-0.058
	(0.053)	(0.073)	(0.060)
$\Delta PRICE^{-}_{t-1}$	-0.609**	-0.667**	-0.322**
	(0.066)	(0.097)	(0.078)
$\Delta PRICE^{-}_{t-2}$	-0.325**	-0.283**	-0.220**
	(0.071)	(0.106)	(0.078)
$\Delta PRICE^{-}_{t-3}$	-0.196**	-0.192	-0.356**
	(0.067)	(0.101)	(0.075)
$\Delta PRICE^{-}_{t-4}$	-0.165**	-0.017	-0.118
	(0.058)	(0.088)	(0.067)
$PRICE_{t-1}$	0.827**	0.717**	0.831**
	(0.038)	(0.058)	(0.029)
TGP_{t-1}	0.170**	0.264**	0.095**
	(0.039)	0.264	(0.029)
MONTHLY DUMMIES	Y	Y	Y
R-SQUARED (PRICE)	0.990	0.986	0.988
NUM. OBS.	545	268	480

Standard errors in parentheses. * Significant at 5% level, ** Significant at 1% level.