Reducing the Contribution of the Power Sector to Ground-Level Ozone Pollution:
An Assessment of Time-Differentiated Pricing of Nitrogen Oxide Emissions

by

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Abstract

Nitrogen oxide (NOx) is a prevalent air pollutant across the United States and a requisite precursor for tropospheric (ground-level) ozone formation. Both pollutants significantly impact human health and welfare, so National Ambient Air Quality Standards (NAAQS) have been established for each. As of 2013, over 100 million people in the U.S. lived in areas with ozone concentrations above the NAAQS.

NOx emissions from the power sector, roughly 12% of total NOx emissions, are and will be significant contributors to ozone concentrations in the U.S. As such, states have reduced peak ozone concentrations through technology-based standards and cap-and-trade programs on NOx emissions from the power sector. These policies have largely treated NOx emissions uniformly. But marginal damages from NOx emissions are greatest on hot sunny days when meteorological conditions favor high ozone formation rates and, consequently, peak ozone concentrations.

This thesis informs what type of policy is the most efficient for reducing peak ozone concentrations on high ozone days by assessing the cost-effectiveness of three policies for reducing NOx emissions on high ozone days. Emissions and costs under a relatively-novel differentiated policy, time-differentiated pricing, are compared for the first time to two currently-implemented undifferentiated policies, cap-and-trade and technology-based standards. Two power systems are studied, Texas and the Mid-Atlantic. A unique two-phase model is developed to capture the short- (redispatching) and long-term (control technology installation) effects of pricing schemes on power plants. The two-phase model dispatches generators with a unit commitment model, which, unlike past studies, captures real-world operational constraints of generators that may strongly influence emissions and costs under time-differentiated pricing. Technology-based standards are simulated via Monte Carlo analysis to capture the uncertain rulemaking process.

For reducing NOx emissions on high ozone days in both power systems, time-differentiated pricing is shown to be the most cost-effective policy with regards to producer and consumer costs. Most emissions reductions are due to substitution of gas- for coal-fired generators, as control technology installations are only observed at very high time-differentiated prices. For reducing summer-wide NOx emissions, undifferentiated pricing is the most cost-effective. In a minority of allocations, technology-based standards also achieve more cost-effective summer-wide reductions than time-differentiated pricing, but such allocations cannot be guaranteed *ex ante*. These results suggest that time-differentiated pricing is the most efficient policy for reducing peak ozone concentrations, depending on ozone formation rates.

Thesis Supervisor: Mort D. Webster
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Table of Contents

List of Figures

List of Tables

Chapter 1: Introduction

Nitrogen oxide (NOx) is a prevalent air pollutant across the United States that is a required precursor for ground-level ozone formation and also contributes to particulate matter. All three pollutants significantly impact human health and welfare and are classified as criteria air pollutants under the Clean Air Act. Because they are criteria pollutants, the U.S. Environmental Protection Agency (EPA) has set National Ambient Air Quality Standards (NAAQS) for each pollutant at a level deemed safe for human health. States in which a criteria air pollutant's concentration exceeds its NAAQS are said to be in nonattainment for that standard. As of 2013, most major metropolitan areas were in nonattainment for the 2008 8-hour ozone standard. Consequently, states in which these areas are located must draft "state implementation plans" (SIPs) that detail how the ozone NAAQS will be "attained".

NOx emissions from the power sector, which account for roughly 12% of total NOx emissions, are and will continue to be significant contributors to ozone concentrations in the U.S. (Fann, Fulcher, & Baker, 2013). As such, over the past decade, the primary mechanisms that eastern states have used to address ozone nonattainment have been NOx cap-and-trade programs on the power sector. The current cap-and-trade program, the Clean Air Interstate Rule (CAIR), is representative of these programs. It allocates annual and summertime NOx emission budgets among eastern states and then ratchets the budgets downwards to reduce aggregate NOx emissions. The proposed successor to the CAIR, the Cross-State Air Pollution Rule (CSAPR), adopts a similar approach to reducing NOx emissions.

Notably, neither program differentiates NOx emissions that occur within the summer based on when they occur. Yet, numerous studies have demonstrated that NOx emissions have highly variable damages depending on when they are emitted in the summer (Levy, Baxter, & Schwartz, 2009; Mauzerall et al., 2005). Of interest to this research, marginal damages from NOx emissions are typically greatest on hot sunny days when meteorological conditions favor high ozone formation rates (Mauzerall et al., 2005). It is also on these days that ozone concentrations typically peak and are mostly likely to exceed the NAAQS.

For states in nonattainment for the ozone NAAQS, therefore, there is a legal and public health rationale for reducing NOx emissions that contribute to peak ozone concentrations. The public health rationale is also relevant to states that are in attainment with the standard. Given these facts, a crucial question is what the most efficient regulatory instrument is for reducing NOx emissions that contribute to peak ozone concentrations.

Two lines of evidence suggest that a time-differentiated market-based instrument would be the most efficient such instrument. First, recent data indicates that the effectiveness of undifferentiated cap-and-trade programs at reducing ozone concentrations has declined in recent years (Martin, 2008; Mauzerall et al., 2005). In part, this could be due to power plants conserving NOx emission permits for use late in the summer, when ozone concentrations typically peak. Second, economic theory predicts that differentiating regulations to account for variability in marginal damages of the regulated pollutant will improve regulatory efficiency (Tietenberg, 2010). Empirical studies have corroborated this theory with respect to NOx emissions (Muller & Mendelsohn, 2009). Given that NOx emissions tend to have the greatest damages when emitted on days when ozone concentrations peak, implementing a time-differentiated policy for reducing NOx emissions on those days would likely be the most efficient policy.

This thesis informs what type of policy is the most efficient for reducing peak ozone concentrations on high ozone days, i.e. for reducing nonattainment with the ozone NAAQS, by assessing the cost-effectiveness of various policies for reducing NOx emissions on high ozone days. It is worth emphasizing that this paper does not directly address policy efficiency, as health benefits are not determined or monetized. Rather, this paper focuses on the cost of emissions reductions under each policy, leaving the calculation of benefits from these reductions to future work.

Three types of policies are assessed in this research. Two of those policies are undifferentiated policies that mirror current regulations on NOx emissions: undifferentiated pricing and technology-based standards. Undifferentiated pricing is used here as a proxy for cap-and-trade, since emissions under the two policies should theoretically be the same when the permit price of the cap-and-trade program is equal to the undifferentiated price. The third policy is a differentiated policy that would assess a price on NOx emissions that occur on forecasted ozone nonattainment days. This policy is modeled as being layered on top of the existing cap-and-trade program, the CAIR, such that emission prices on non-high ozone days are equal to current undifferentiated emissions prices under the CAIR. Consequently, NOx emission pricing under this time-differentiated policy can be thought of as a step function that increases from a baseline to higher price for 24 hours on high ozone days.

Two prior studies demonstrated such a time-differentiated pricing scheme can achieve significant reductions in NOx emissions on high ozone days without threatening grid reliability (Bharvirkar, Burtraw, & Krupnick, 2004; Sun et al., 2012). But both studies ignored important real-world operational constraints on electricity generators by using a simplified power system model, and studied the same power system. Additionally, neither study compared the time-differentiated pricing scheme to an undifferentiated price or to realistic technology-based standards.

This work builds on these prior studies by comparing the cost-effectiveness of the proposed time-differentiated pricing scheme to undifferentiated pricing and technology-based standards for the first time. System-wide costs and emissions are compared across various strengths of the three types of policies for the entire summer as well as on high ozone days to explore tradeoffs associated with each policy approach. Costs and emissions under technology-based standards are calculated via Monte Carlo analysis, wherein control technology mandates are randomly allocated to generators to account for the uncertain nature of the policy implementation process.

Emissions and costs are determined with a two-phase model that captures the short- and long-term responses of generators to NOx emission prices. In the short-term, substitution between generators occurs as generators with lower NOx emissions supplant electricity with generators with higher NOx emissions. In the long-term, generators can install control technologies, specifically Selective Catalytic Reduction (SCR), to minimize their NOx emissions, mitigating the impact of the price on operational costs. SCR installations are determined under Nash equilibrium, wherein generators are allowed to reconsider installation decisions in light of other generators' decisions. This novel application of Nash equilibrium captures key feedbacks of control technology installations on system-wide electricity prices and installation decisions of other generators.

The two-phase model relies on a detailed power system (unit commitment) model that determines hourly power output at each generator while accounting for generators' real-world operational constraints. The unit commitment model better captures power system operations, and therefore provides a more realistic estimate of emissions and costs under differentiated pricing, than those used in prior studies. Analyses are conducted in two power systems, Texas and the Mid-Atlantic, that have different fuel mixes in order to facilitate cross-system comparisons.

Of the three types of policies analyzed here, the time-differentiated pricing scheme is the most cost-effective for reducing NOx emissions on high ozone days in both power systems. This is true in regards to producer and consumer costs. Moreover, as found in prior studies that used a simpler power system model, the emission reductions achievable with the differentiated pricing scheme are significant – over 50% in both systems – and sufficient flexibility in both power systems exists to maintain grid reliability and meet demand. Most emission reductions and cost increases stem from substitution of gas- for coal-fired generation, even when SCR is installed at some generators. SCR installations only occur at the higher time-differentiated prices tested.

Time-differentiated pricing is not a panacea, though, as it is not well suited to reducing summer-wide NOx emissions. Compared to technology-based standards, time-differentiated pricing yields more cost-effective emission reductions in most cases. There are some possible allocations of SCR under a technology-based standard that can achieve more cost-effective summer-wide reductions than a time-differentiated price, but given the uncertain nature of the rulemaking process, it's highly unlikely such allocations could be guaranteed *ex ante*. Thus, while technology-based standards can theoretically achieve more cost-effective summer-wide emissions reductions, time-differentiated pricing would likely be more cost-effective in reality. However, time-differentiated pricing can only reduce summer-wide NOx emissions to a limited extent over the summer, as it achieves little to no additional emission reductions on non-high ozone days. Consequently, depending on the desired level of summer-wide NOx emission reductions, a technology-based standard may be superior to time-differentiated pricing. But both types of policies are shown to be far less cost-effective than an undifferentiated price for summer-wide NOx emission reductions.

Several differences emerge between PJM, a coal-dominated system, and ERCOT, which has a much higher share of low-cost baseload gas-fired generation. First, cost savings under the differentiated pricing scheme are greater in ERCOT than in PJM, suggesting systems with less coal and more low-carbon sources of electricity, like wind and gas, would benefit more from a differentiated regulatory approach. At the same time, a greater share of emissions on high ozone days can be reduced via differentiated regulation in PJM. Regarding SCR installations, a higher capacity was installed in PJM at a lower differentiated price than in ERCOT.

These results suggest that the time-differentiated pricing scheme is the best policy instrument for abating ozone concentrations on high ozone days, i.e. for reducing ozone nonattainment. If true, this means that the EPA can achieve substantial benefits from augmenting its current strategy of tightening undifferentiated cap-and-trade programs to a differentiated regulation as proposed here for reducing ozone nonattainment. However, this conclusion cannot be firmly established because of the highly complex interaction between NOx emissions and ozone concentrations. In fact, it is possible that decreasing NOx emissions would actually increase ozone concentrations in parts of the regions studied here, such that greater emission reductions would be worse. Additional complexity arises from the difference in when and where emission reductions occur between the types of policies assessed here. Of particular note, the differentiated pricing scheme reduces emissions almost exclusively on high ozone days, whereas the other policies reduce emissions further in advance and on those days. Due to all of these factors, a clear link cannot be drawn between changes in emissions and resultant ozone concentrations and public health effects without conducting air quality modeling. As an extension of this research, though, air quality modeling with the Comprehensive Air Quality Model with Extensions (CAMx) will be done by colleagues at the University of Texas in Austin.

Results of this research also demonstrate that the time-differentiated pricing scheme is not a substitute for undifferentiated pricing for summer-wide ozone concentration reductions. As such, the differentiated scheme proposed here would be best suited as an additional program layered on top of existing undifferentiated programs. Implementing it in this manner would take advantage of existing trading and communications infrastructure already in use by current regulations. Implementation of the differentiated scheme is also legally justifiable under the Clean Air Act and could be done using existing ozone forecasting technology.

Chapter 2: Sources and Primary and Second Impacts of Nitrogen Oxides

Nitrogen oxides (NOx) are a class of reactive air pollutants that are emitted across the U.S. from stationary and mobile sources. NOx emissions have direct impacts on public health and welfare, but also incur indirect impacts through the formation of tropospheric ozone and particulate matter. This chapter begins by surveying the sources and direct, or primary, impacts on human health from NOx. Secondary health impacts of NOx emissions via contribution to ozone and particulate matter formation are then presented. Finally, the state of NOx and ozone pollution across the U.S. is given along with the contribution of the power sector to ozone and particulate matter pollution.

2.1: Nitrogen Oxides

Nitrogen oxides (NOx) are a class of commonly emitted, reactive air pollutants that include nitrogen dioxide (NO_2), nitric oxide (NO) and nitrous acid (HNO_3). NOx directly impact public health and welfare and the environment. These direct impacts are referred to as primary impacts. NOx emissions also incur secondary impacts by contributing to the formation of tropospheric ozone (O_3) and particulate matter (PM). Because of their prevalence and risk of harm to human health and welfare, NOx is regulated as a criteria pollutant under the Clean Air Act.

2.1.1: Sources of NOx

Three sectors of the U.S. economy account for over 80% of total annual NOx emissions (Table 1). The electricity sector is the largest stationary source of NOx emissions. Although it constituted only 12% of NOx emissions in 2013, until 2009 it typically accounted for roughly 18% of annual NOx emissions (U.S. Environmental Protection Agency, 2011d). This declining share has been driven by declining NOx emissions from the sector; since 2009, NOx emissions from electricity generation declined from 2,700 to 1,600 thousand tons, while total NOx emissions declined from 15,500 to 12,600 thousand tons (U.S. Environmental Protection Agency, 2011d).

Table 1: Proportion of total 2013 NOx emissions from three largest emitting sectors (U.S. Environmental Protection Agency, 2011d).

Source Category	Percent Annual Emissions (%)
Electricity Fuel Combustion	12
Industrial Fuel Combustion	10
Vehicles (Highway and off-highway)	61

The decrease in power sector NOx emissions has been driven by a reduction in coal-fired power generation, which have much higher NOx emissions rates than any other fuel (Jaramillo, Griffin, & Matthews, 2007). As such, coal power plants emit the vast majority of NOx from the power sector. In 2013, for instance, coal plants emitted 92% of NOx emissions from the power sector based on data from the Acid Rain Program (U.S. Environmental Protection Agency, 2014) while only generating 40% of that year's electricity[1] (U.S. Energy Information Administration, 2014).

[1] Specifically, from January to October of 2013, the last month for which data was available at the time of writing.

Most NOx emitted from power plants is thermal NOx, meaning it's produced when nitrogen in air dissociates at high temperatures and then reacts with oxygen (Richards, Weiland, & Strakey, 2006). As such, multiple factors affect NOx formation during power generation, including the combustion temperature and the residence time of combusted air. Plant heat rates, or how efficiently they convert input fuel to electricity, also affect emissions rates, and coal plants are on average less efficient than other baseload NOx-emitting power sources like natural gas.

Once emitted, the lifetime of NOx in the troposphere is roughly one day, depending on meteorological conditions and proximity to the surface (Jacob, 1999a; U.S. Environmental Protection Agency, 2006; Zhang, Tie, & Bond, 2003). This means that NOx emissions from a power plant will incur primary impacts or contribute to the formation of secondary pollutants within a few days of being emitted, depending on local conditions (Bharvirkar et al., 2004).

2.1.2: Primary Impacts of NOx

Exposure to NOx has been linked to adverse respiratory effects, including airway inflammation and exacerbated symptoms in asthmatics. These effects, in turn, lead to increased hospital and emergency room visits. The vast majority of impacts from NOx emissions, though, are secondary impacts through tropospheric ozone and particulate matter formation.

2.2: Tropospheric Ozone

Ozone can be found in the stratosphere – the upper layer of the atmosphere – and the troposphere – the lower layer of the atmosphere. While stratospheric ozone provides immense health benefits by blocking ultraviolet rays from reaching the Earth's surface, tropospheric (ground-level) ozone has a number of harmful health impacts. NOx emissions contribute to the formation of tropospheric ozone. Like NOx, tropospheric ozone is a criteria air pollutant under the Clean Air Act because of its prevalence and threat to human health and welfare.

2.2.1: Formation of Tropospheric Ozone

Ozone is not directly emitted, but rather forms via chemical reaction. The vast majority of tropospheric ozone – roughly 80-88% – is formed via reactions in the troposphere; the remainder is transported from the stratosphere (Jacob, 1999a). However, in areas with high ozone levels, such as urban areas, the fraction of tropospheric ozone originating from the stratosphere is likely much smaller than that amount.

Ozone is formed via reaction between atomic and molecular oxygen:

$$O + O_2 \rightarrow O_3$$

In the troposphere, the only significant source of O is from the dissociation of NO_2 in the presence of sunlight:

$$NO_2 + h\upsilon \rightarrow NO + O$$

In other words, tropospheric ozone can only form in the presence of NOx, specifically NO_2, and sunlight. Yet, ozone can also generate NO_2 through reaction with NO, another NOx compound:

$$NO + O_3 \rightarrow NO_2 + O_2$$

Thus, concentrations of ozone are determined by the relative rate constants of these two reactions, as ozone is produced and consumed in each (Seinfeld, 1989).

NO_2 can also be formed via reaction with hydrocarbons, including volatile organic compounds (VOCs). For instance, a peroxy radical can be created when carbon monoxide[2] is oxidized by a hydroxyl group in the presence of oxygen:

$$CO + OH \rightarrow CO_2 + HO_2$$

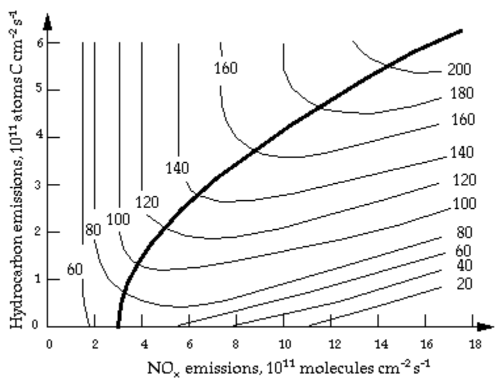
and can then react with NO to produce NO_2:

$$HO_2 + NO \rightarrow OH + NO_2$$

thereby increasing concentrations of NO_2 (Jacob, 1999a). Increased NO_2, in turn, leads to increased O_3 formation. Thus, ozone formation is driven by the presence of VOCs as well as NOx.

The relationship between NOx, VOC and ozone concentrations is complex (Figure 1) because ozone formation does not increase linearly with increases in either pollutant. Rather, two regimes of ozone formation exist in which ozone levels will respond differently to changes in NOx or VOCs. In a NOx-limited regime (above the thick line in Figure 1), increasing NOx levels increases ozone, but reducing hydrocarbon emissions has little to no effect on ozone. Conversely, in a hydrocarbon-limited regime (below the thick line in Figure 1), ozone levels decrease with decreasing VOC concentrations, but not with decreasing NOx concentrations. In general, rural and suburban areas are usually NOx-limited while downtown urban areas are VOC-limited (U.S. Environmental Protection Agency, 2006).

Figure 1: Ozone concentrations (lines) as a function of NOx and hydrocarbon emissions. The thick line separates two ozone-forming regimes: above is NOx-limited and below is VOC-limited (Jacob, 1999b).



Given the complex relationship between ambient NOx and ozone concentrations, it is extremely difficult to predict how ozone concentrations will react to changes in NOx emissions from power plants. However, (Fann et al., 2013) show that the power sector will continue to be one of the largest contributing sectors of the U.S. economy to ozone concentrations in 2016. This suggests that plants will continue to contribute to ozone formation through NOx emissions for years to come, and that reducing those NOx emissions will yield reductions in ozone concentrations.

Ozone concentrations follow a diurnal cycle. Concentrations peak during the day as ozone is formed in the presence of sunlight. At night, ozone concentrations decline as ozone is depleted. Ozone depletion is driven by titration with nitric oxide, dry deposition, and transport via wind (Bloomer, Vinnikov, & Dickerson, 2010; Fowler et al., 2008; U.S. Environmental Protection Agency Office of Air Quality, 1999; World Health Organization, 2000).

[2] Volatile organic compounds (VOCs) can also be oxidized in order to form peroxy radicals.

Ozone formation is strongly impacted by meteorological conditions. Specifically, ozone formation is greatest on hot sunny days when sunlight is strongest (Fowler et al., 2008; Mauzerall et al., 2005; Tong & Muller, 2006). As such, ozone formation can strongly vary over the course of only a few days as meteorological conditions change (Mauzerall et al., 2005). It also varies spatially, given the complex relation between ozone formation and NOx and VOC concentrations (Mauzerall et al., 2005; Tong & Muller, 2006). The spatial and temporal variability of ozone formation is discussed at greater length in Section 3.2.

Tropospheric ozone has a lifetime of roughly 1-2 days in the boundary layer, i.e. at the planet surface (Fowler et al., 2008). Its lifetime is longer, at around 20 days, higher in the troposphere (Stevenson et al., 2006).

2.2.2: Impacts of Tropospheric Ozone

Ground-level ozone has a number of deleterious effects. Ozone is the primary component of smog, which can significantly impair visibility. Ozone also has numerous adverse health effects due to its ability to oxidize, or damage, biological tissue (Jacob, 1999a). Based on epidemiological and other studies, short-term ozone exposure has been robustly linked to various negative effects on respiratory health, including increased hospital and emergency room visits for respiratory complaints (Berman et al., 2012; Fann et al., 2012; Hubbell et al., 2005; U.S. Environmental Protection Agency, 2006). Short- and long-term ozone exposure has also been linked strongly to increased risk of mortality and respiratory disease (Berman et al., 2012; Fann et al., 2012). Various groups are more susceptible to impacts from ozone pollution, including children (under 18) and elderly and individuals with respiratory ailments like asthma (U.S. Environmental Protection Agency, 2006).

Health effects of ozone scale log-linearly with ozone concentrations, meaning health impacts are greater at higher ozone concentrations (Berman et al., 2012; Fann et al., 2012). Consequently, the greatest health impacts from ozone exposure would be expected to occur on high-ozone days when ozone concentrations are greatest. The nonlinear relationship between ozone concentrations and health impacts is captured in a health impact function, which relates changes in health outcomes to changes in pollutant concentrations via the following relationship:

$$\Delta y = y_0 \cdot (e^{\beta \cdot \Delta x} - 1)$$

where Δy is the change in health effects; y_0 is the baseline incidence of health effects; β is the health effect estimate, which is drawn from epidemiological studies; and Δx is the change in pollutant concentrations (Berman et al., 2012; Fann et al., 2012). Multiplying the above equation by population yields aggregate health effects.

As of yet, no threshold effect has been established for health effects from ozone exposure (Correia et al., 2013; Hubbell et al., 2005; U.S. Environmental Protection Agency, 2006), meaning there is no clear ozone concentration below which impacts are not incurred. Moreover, some data suggests that even if a threshold existed, it would be near the lower limits of ambient ozone concentrations extant in the U.S. (Bell, Peng, & Dominici, 2006; U.S. Environmental Protection Agency, 2006).

2.3: Particulate Matter

2.3.1: Formation of Particulate Matter

Particulate matter (PM) can be directly emitted ("primary PM") or formed via chemical reaction ("secondary PM") (U.S. Environmental Protection Agency, 2012d). NOx contributes to secondary PM via reaction with ammonia, moisture or other compounds in the atmosphere.

Nationwide, nitrates, which are derived from NOx, are the dominant secondary PM species along with sulfates and ammonium salts (Ansari & Pandis, 1998).

However, the composition and precursors of PM vary greatly across regions in the U.S. (U.S. Environmental Protection Agency, 2012d). For one, regions within the U.S. diverge greatly in the share of atmospheric PM that is primary versus secondary in nature. Regions also differ greatly in how much PM is nitrate, i.e. derived from NOx. For instance, in the Midwest, nitrates make up a small part of PM, whereas in Southern California nitrates account for a majority of PM (U.S. Environmental Protection Agency, 2012d). PM composition also varies seasonally – nitrate PM is more prevalent in cool months and nearly non-existent in warm months across most of the nation.

2.3.2: Impacts of Particulate Matter

Exposure to PM has significant adverse health effects. Chronic exposure to PM2.5, or PM of a diameter less than 2.5 microns, has been robustly linked to increased risk of premature death (Fann et al., 2012; U.S. Environmental Protection Agency, 2010). Short- and long-term exposure has also been linked to respiratory and cardiovascular ailments, exacerbated asthma symptoms, and nonfatal heart attacks (Fann et al., 2012; U.S. Environmental Protection Agency, 2010). Suggestive links exist between PM2.5 exposure and reproductive and developmental effects, cancer, mutagenicity and genotoxicity (U.S. Environmental Protection Agency, 2010). Weaker evidence of similar impacts from exposure to PM10, or PM of a diameter less than 10 microns, also exists (U.S. Environmental Protection Agency, 2010). Like ozone, health impacts from PM exposure are non-linear in nature, as expressed by health impact functions.

2.4: Current State of NOx and Ozone Pollution

Despite reductions in recent decades (discussed in Chapter 3), many cities across the U.S. still suffer from harmful levels of NOx and ozone pollution, including peak ozone concentrations in excess of federal ambient concentration standards that are set at levels to protect human health and welfare. These ambient concentration standards, called the National Ambient Air Quality Standards, are set for all six criteria pollutants under the Clean Air Act (CAA), of which NOx and ozone are two. Primary and secondary NAAQS are set to protect human health and welfare, respectively; the current ozone primary and secondary NAAQS are set such that the annual fourth-highest daily maximum 8-hour ozone concentration averaged over 3 years is 75 parts per billion (ppb). Two NOx primary NAAQS standards exist: an annual average for NO_2 at 53 ppb, and a 3-year average of the 98^{th} percentile of 1-hour NO_2 concentrations at 100 ppb.

Compliance with NAAQS standards provides a good snapshot of the state of NOx and ozone pollution across the U.S. Areas in which ozone and NOx concentrations are below the NAAQS standards are said to be in attainment, whereas those above, or in "exceedance", of the NAAQS standards are said to be in nonattainment. Thus, because the ozone NAAQS is based on the highest annual ozone concentrations, nonattainment for the ozone NAAQS indicates that high-ozone days remain a problem for the city or area.

No areas in the U.S. are currently in nonattainment for the NOx NAAQS standard (U.S. Environmental Protection Agency, 2013c). Furthermore, only one area, which consists of Los Angeles, Orange, Riverside and San Bernardino counties, is in maintenance for the standard, meaning it recently achieved attainment with the standard.

Conversely, 227 counties, which contain a total population of over 120 million, are in nonattainment for the ozone NAAQS (**Figure 2: Areas in nonattainment (shaded) for the 8-**

hour ozone NAAQS (2008 Figure 2). Furthermore, of those counties in nonattainment, 23 are in "serious", "severe" or "extreme" nonattainment for the ozone standard, meaning ozone concentrations in those counties exceed the ozone standard by 33% or more. In other words, more than a third of Americans live in areas where ozone levels exceed those concentrations deemed necessary to protect public health. These exceedances occur on high-ozone days, underscoring the ongoing challenge many cities face in abating high-ozone days.

Ozone levels in exceedance of the NAAQS, i.e. on high-ozone days, exact a significant toll in lives and wellbeing. (Hubbell et al., 2005) estimated that bringing ozone levels across the U.S. into attainment with the NAAQS would save roughly 1,000 lives and prevent thousands of hospital visits each year. (Berman et al., 2012) found similarly-large health benefits of reducing ozone concentrations to the NAAQS standard. According to this analysis, reducing only ozone concentrations above the standard to 75 ppb would avoid roughly 400 to 800 deaths, along with millions of acute respiratory symptoms and lost school days, per year.

Figure 2: Areas in nonattainment (shaded) for the 8-hour ozone NAAQS (2008 standard) as of December 2013 (U.S. Environmental Protection Agency, 2013a). Partially shaded counties indicate that only part of the country is in nonattainment.

Chapter 3: Historic and Current Regulation of NOx

NOx has been regulated in the United States as a criteria air pollutant since the passage of the Clean Air Act in 1970. Regulation of NOx is a classic example of correcting for a negative externality, as the cost of impacts from NOx emissions on humans and the environment would otherwise not be priced by the economy (Viscusi, Vernon, & Harrington, 2005). This chapter begins with a survey of past and present regulations on NOx emissions from the power sector in the U.S. It then reviews the highly differentiated nature across space and time of the impacts from NOx emissions. Given this discussion, two justifications for preferential reduction of NOx emissions that contribute to high ozone concentrations are given. The tradeoffs between a market-based approach and technology-based standard are then explored for achieving this goal, and two lines of evidence are finally presented that suggest the goal would be best achieved with a time-differentiated market-based instrument.

3.1: Survey of Regulations on NOx Emissions from Power Plants in the U.S.

3.1.1: Initial Regulation: The Clean Air Act and Subsequent Amendments

Regulation of NOx, ozone and other air pollution truly began in 1970 with the passage of the Clean Air Act (CAA). The Act included many measures aimed at reducing air pollution. One such measure was the establishment of primary and secondary National Ambient Air Quality Standards (NAAQS) for "criteria pollutants", i.e. any prevalent air pollutants that endanger the public health and welfare (42 U.S. Code §§ 7408-7409). Six air pollutants were selected as criteria pollutants, including NOx, ozone and PM. For these pollutants, the EPA sets primary NAAQS that specify an ambient concentration of the pollutant that is necessary to protect the public health (42 U.S.C. § 7409(b)(1)). Secondary NAAQS may also be set to protect the public welfare (42 U.S.C. § 7409(b)(2)). States must comply with the NAAQS by drafting state implementation plans, or SIPs, that dictate either how they will reduce ambient concentrations of a given criteria pollutant if concentrations of that pollutant exceed the NAAQS, or how concentrations will be maintained below the NAAQS if they do not exceed it (42 U.S.C. § 7410). These plans must include how emissions will be reduced in order to meet or continue to comply with the NAAQS, if necessary (42 U.S.C. § 7410(a)(2)(A)).

In addition to establishing a NAAQS for NOx and requiring states to implement strategies for reducing NOx emissions, the CAA also set performance-based standards on NOx emissions from new sources. These standards, which were called New Source Performance Standards (NSPS) and further revised in the CAA Amendments of 1977, established an upper limit on NOx emissions rates at new facilities. Specifically, the NSPS essentially require "best available control technology" (BACT) at new plants in areas of attainment for the ozone NAAQS. For new facilities in areas in nonattainment for the ozone NAAQS, more stringent "lowest achievable emissions reduction technology" (LAERT) was required along with emission offsets through reductions at existing sources (Burtraw & Evans, 2003; Swift, 2001).

In 1990, NOx emissions were regulated for the first time at existing sources. These regulations took the form of a nationwide prescriptive approach and a regional trading program. The prescriptive approach, established under Title IV, set emissions rate limits, as the original CAA did. These new limits were set at emission levels that would be attained with "reasonably achievable control technology" (RACT) (U.S. Environmental Protection Agency, 2013b), which in this case meant low-NOx burners or other combustion modifications. The facilities affected by these standards accounted for roughly 85% of the electricity sector's NOx emissions. From 1990 to 2001, emissions from these sources fell by 26%, even as electricity generation from coal grew 16% (Burtraw & Evans, 2003).

At the same time, a regional cap-and-trade program, called the Ozone Transport Commission (OTC) NOx Budget Program (NBP), was established for NOx in the Northeast U.S. Under this program, a region-wide budget was set for NOx emissions during the summer months (May 1 to September 31), and trading was then permitted between affected utilities (Burtraw & Evans, 2003). The OTC was established to combat cross-state pollution of NOx, wherein NOx is transported via wind from upwind to downwind states where it forms ozone and particulate matter. Northeastern states recognized that they were unable to fully address ozone pollution in their own state because at least part of their pollution was actually emitted elsewhere and so was outside their jurisdiction. The region-wide emissions cap was meant to reduce this interstate transport. The NBP was subsequently replaced by the Clean Air Interstate Rule, discussed next.

3.1.2: Recent Cap-and-Trade Programs: The Clean Air Interstate Rule and Cross-State Air Pollution Rule

Over the last decade, the EPA promulgated two cap-and-trade programs in an effort to further reduce power plant emissions and interstate transport of NOx: the Clean Air Interstate Rule (CAIR) in 2005 (U.S. Environmental Protection Agency, 2005), which remains in effect, and the Cross State Air Pollution Rule (CSAPR) in 2011 (U.S. Environmental Protection Agency, 2011b). Both regulations aim to reduce NOx emissions in order to reduce nonattainment with the ozone and particulate matter NAAQS. (The CAIR and CSAPR also set caps on SO_2 emissions, which also contribute to particulate matter.)

The CAIR and CSAPR aim to reduce NOx emissions in the eastern half of the U.S., including Texas. Each program sets region-wide annual and "ozone season" emission budgets that are ratcheted down after a period of time. In the CAIR, for instance, emission budgets entered into force in 2009 and will be ratcheted down in 2015. At its most stringent, the CAIR will require a 60% reduction in region-wide NO_x emissions, respectively, from power plants from 2003 levels by 2015. (The CAIR and CSAPR use different methods to allocate region-wide budgets among states.)

The CAIR and CSAPR are similar to the NBP in that they also would reduce nonattainment with the ozone NAAQS by reducing "significant contributions" to downwind nonattainment by upwind sources, i.e. by reducing interstate transport. "Significant contribution" is strictly defined in the "good neighbor" provision of the Clean Air Act as the amount of pollution an upwind state contributes to a downwind state's air pollution levels in excess of the NAAQS for that pollutant.[3] In fact, the D.C. Circuit Court of Appeals struck down the CAIR in 2008 (State of North Carolina, 2008) and CSAPR (EME Homer, 2012) in 2012 on the basis that both proposed rules violated the "good neighbor" provision. Specifically, the Court found that neither the CAIR nor CSAPR ensured that emission reductions they required were less than or equal to a state's "significant contribution" to nonattainment in downwind states. On appeal, the Court allowed EPA to implement the CAIR until it could be replaced by another program. It did not give the same leeway to the EPA for implementing CSAPR, though. As such, the EPA

[3] For instance, if a downwind state exceeds the ozone NAAQS by one unit of pollution, and an upwind state contributes five units of pollution to that state's air quality, the upwind state's "significant contribution" would only be one unit because it is limited to the amount the downwind wind state exceeds the relevant NAAQS. If, on the other hand, the downwind state exceeded the ozone NAAQS by six units of pollution, the "significant contribution" of the upwind state would be five units of pollution. See (EME Homer, 2012).

appealed the ruling on CSAPR to the Supreme Court, where oral arguments were held on December 17, 2013. At the time of this writing, though, no decision had been issued, and the future of the CSAPR remains in limbo.

Despite the uncertain future of CSAPR, the EPA's ongoing defense of the rule suggests that the agency will continue to use undifferentiated cap-and-trade programs to reduce non-compliance with the ozone NAAQS over the next five to ten years, at least. Crucially, though, any undifferentiated cap-and-trade program like the CSAPR would not differentiate NOx emissions temporally beyond whether they occur between May and September. In other words, in all likelihood, the EPA will likely continue to treat all NOx emissions in the summer as equivalent.

3.2: Spatial and Temporal Variability of Impacts from NOx Emissions

Contrary to their treatment under current regulations, NOx emissions have highly heterogeneous impacts across space and time, including within the summer (Levy et al., 2009; Martin, 2008; Mauzerall et al., 2005; Muller, 2011; Tong & Muller, 2006). NOx is a non-uniformly mixed pollutant (Fowlie & Muller, 2013), meaning it does not evenly disperse once emitted. As such, where and when NOx emissions occur affects damages caused by those emissions through primary and secondary impacts.

At root, primary impacts of NOx are driven by atmospheric concentrations, the size of the exposed population and the susceptibility of that population (Muller, 2011). Atmospheric concentrations of NOx to which populations are exposed are affected by factors like atmospheric conditions and concentrations of other gases and particles that vary across space and time (Levy et al., 2009; Muller, 2011). At any given location or city, for instance, different weather from one day to the next may lead to differing impacts from identical NOx emissions from a given source . The location of the power plant and its proximity to the exposed population will also affect its damages (Levy et al., 2009).

Secondary impacts of NOx emissions, i.e. impacts from PM and ozone that form from NOx, are also highly spatially and temporally variable. This variability arises from the highly variable formation of PM and ozone based on chemical processes and meteorological conditions. PM formation varies across ambient concentrations of ammonia (NH_3), nitrates (products of NOx) and sulfates (products of SO_2) (Ansari & Pandis, 1998). Thus, identical emissions of NOx may have differing results on PM formation depending on ambient concentrations of other gases. PM formation also varies with temperature (Ansari & Pandis, 1998). Ozone formation is driven by meteorological conditions as well, particularly temperature and sunlight (Mauzerall et al., 2005; Tong & Muller, 2006). It also varies according to concentrations of other gases and particles, such as VOCs, as previously discussed (Jacob, 1999a).

Multiple studies have demonstrated the high spatial and temporal variability of damages from power plant NOx emissions using air quality models. (Levy et al., 2009) examined the spatial variability of health-related damages of emissions from coal-fired power plants using a reduced-form air quality model. They found primary and secondary impacts from NOx were highly variable among power plants, largely due to differing population exposures. Multiple factors affected population exposure in their model, including the location of the power plant (e.g., in the Great Plains versus the East Coast). Atmospheric conditions, which affect pollutant accumulation and ozone and PM formation, also affected the variability in damages. Damages for a single ton of NOx emissions ranged from \$0.5 to \$15,000. Secondary PM2.5 impacts, which they also estimated, were highly spatially differentiated as well. (Correia et al., 2013) also documented high spatial variation in the effect of PM2.5 on life expectancy.

(Muller, 2011) also looked at the spatial variability of damages of various pollutants from coal power plants. Muller used Monte Carlo analysis with an Integrated Assessment Model, rather than a full air quality model, to link power plant emissions to damages. Damages from NOx emissions varied highly between intra- and inter-state generators. Variability in damages stemmed from primary and secondary impacts. Variability in NOx damages was extremely high: damage standard error was nearly twice the mean value. In some locations, damages in urban areas from nearby incremental NOx emissions were even negative, meaning they would have positive health effects, because they reduce ozone concentrations through titration.

(Tong & Muller, 2006) examines the spatial variation of ozone formation and impacts from NOx emissions. Using an Integrated Assessment Model that includes a detailed air quality model, the Community Multiscale Air Quality model, Tong and Muller analyze the impacts of an identical increase in NOx emissions from nine rural and urban counties in and around Atlanta, Georgia. They find that human exposure to ozone varies by roughly 250% in magnitude depending on where NOx emissions are increased, e.g. in Atlanta versus in upwind rural areas. Moreover, in some locations such as in Atlanta, increased NOx emissions actually reduce ozone exposure due to titration, whereas increased emissions in upwind rural counties significantly increase ozone exposure.

Complementing (Tong & Muller, 2006), (Mauzerall et al., 2005) depicts the high spatial and temporal variation in secondary NOx impacts through ozone formation. Both papers take a similar approach to measuring the variability in secondary NOx impacts, in that they quantify the impacts of an identical NOx emission profile shifted across space or time. Mauzerall et al. show that for identical NOx emissions just five days apart in July, consequent ozone concentrations are about 50% greater during a warm period than a cool one. This disparity in impacts arises from increased formation of ozone from NOx in hot sunny weather, and highlights the highly-variable nature of NOx impacts through ozone formation across a small period of time – much smaller than the differentiation of the "ozone season", i.e. May to September, by current regulations. Mauzerall et al. also demonstrate that shifting identical emissions geographically (from Maryland to North Carolina) while controlling for temperature can result in nearly twice as great maximum ozone concentrations. Impacts of different resulting ozone concentrations, though, are also driven by population exposure, meaning increases in ozone concentrations do not correlate directly with increases in health impacts.

These studies demonstrate that damages from NOx emissions vary greatly depending on when and where the emissions occur. This research grapples with one component of the temporal variability: impacts from summertime NOx emissions tend to be greatest on hot sunny days, when meteorological conditions favor ozone formation and consequent nonattainment with the ozone NAAQS (Mauzerall et al., 2005).

3.3: Justification for Further Reducing NOx Emissions that Contribute More to High Ozone Concentrations

Based on the above discussion, two rationales exist for achieving greater reductions in NOx emissions that contribute more to peak ozone concentrations. The first is that states must comply with the ozone NAAQS, which requires reducing (if the state is in nonattainment) or maintaining (if the state is in attainment) peak ozone concentrations. To do so, states must submit State Implementation Plans that lay out how the state will reduce NOx emissions[4] in order to attain the NAAQS. These reductions often come at least in part from the power sector given that it is the largest stationary source of NOx emissions, is made up of large point sources that are easier to regulate than diffuse mobile sources like vehicles, and is a large contributor to nonattainment with the ozone NAAQS (Fann et al., 2012). Second, because NOx emissions tend to have the greatest impacts on days when ozone concentrations peak, further reducing NOx emissions that contribute more to peak ozone concentrations would yield greater health benefits than other NOx emission reductions.

Thus, legal and public health rationales exist for further reducing NOx emissions that contribute more to peak ozone concentrations. The question, then, is how these reductions could be achieved best, i.e. most efficiently. The remainder of this chapter discusses the merits of various instruments for achieving these reductions, and concludes that a time-differentiated market-based instrument would likely be the most efficient option.

3.4: Technology- and Performance-Based Standards versus Market-Based Instruments

Regulations on NOx emissions from the power sector have used technology- or performance-based standards or market-based instruments. Technology- and performance-based standards, sometimes referred to as “command-and-control” regulations, require plants to either install a particular technology or meet a particular performance standard, e.g. an emissions rate. Such standards have been used to regulate NOx emissions since the passage of the Clean Air Act in 1970. Market-based instruments, on the other hand, encompass taxes and cap-and-trade programs. Market-based instruments provide greater flexibility in compliance, in that they do not mandate particular actions be taken at individual plants. Rather, they set economy-wide standards (such as via a cap-and-trade program) or send price signals (such as via a tax), and allow the market to determine the best way to meet or accommodate those standards or signals.

Most studies have found market-based instruments to be more economically efficient than technology- or performance-based standards for reducing emissions from the power sector (Carlson et al., 2000; Ellerman, 2003; Schmalensee & Stavins, 2012). Much of the efficiency gains arise from lower costs of compliance due to greater flexibility under market-based instruments. Because the marginal cost of abatement varies among plants, technology and performance requirements on each plant can result in very high compliance costs at certain plants. Conversely, under market-based approaches, reductions can be achieved at plants with the lowest marginal cost of abatement, lowering aggregate compliance costs (Carlson et al., 2000; Hahn & Stavins, 1991; Schmalensee & Stavins, 2012). Other factors also make market-based instruments more efficient. For instance, by not prescribing a particular emission reduction mechanism, market-based approaches allow plants to take advantage of unforeseen developments that allow for cheaper compliance. An excellent illustration of this occurred under the Acid Rain Program, which regulated SO_2 emissions from the power sector, when railroad deregulation led to reduced costs and wider availability of low-sulfur coal, which proved to be a cheap method of reducing SO_2 emissions (Ellerman & Montero, 1998).

Another shortcoming of technology- and performance-based standards is the messy process by which the standards are formulated and allocated. Such standards are developed and implemented through administrative processes that Presidential administrations and Congress can influence (McCubbins, Noll, & Weingast, 1987). Outside interest groups can exert their power through these channels in order to ensure the standards are crafted in their favor. Interest

[4] In rare instances, SIPs may address ozone concentrations via means other than reducing NOx emissions.

groups, especially industry and trade associations, are also allowed extensive input into the crafting and implementation of standards (Stavins, 1998). (While input is also solicited from these groups during the development of market-based instruments, such instruments operate on the entire economy so there is less ability for individual actors to benefit from specific alterations to the rule (Stavins, 1998)). Consequently, technology- and performance-based standards are often far more stringent on new than existing sources (Keohane, Revesz, & Stavins, 1998; Swift, 2001). Existing sources are even sometimes "grandfathered" under new regulations, meaning they do not have to comply with those regulations (Keohane, Revesz, & Stavins, 1998). Another result of this messy process is that it is difficult, if not impossible, to predict *ex ante* which plants a technology- or performance-based standard would affect.

The above discussion demonstrates the many advantages of market-based instruments compared to technology- or performance-based standards. In light of this fact, the next section focuses on the use of market-based instruments in reducing NOx emissions, specifically on high ozone days.

3.5: Market-Based Instruments for Reducing NOx Emissions: Evidence for Efficiency Gains from Temporal Differentiation

As discussed above, market-based instruments have historically regulated NOx emissions through a largely-undifferentiated approach. That is, these instruments have treated NOx emissions the same, with the exception of differentiating between NOx emissions that occur during the summer. Yet, two lines of evidence suggest that market-based instruments that differentiate NOx emissions based on when they occur *within* the summer – in particular, whether they occur on high ozone days – could achieve more efficient reductions of NOx emissions on high ozone days and improve overall regulatory efficiency. First, economic theory predicts, and empirical studies confirm, that regulating heterogeneous pollutants with differentiated instruments improves efficiency. Second, data suggests that NOx emission reductions obtained under undifferentiated cap-and-trade programs are yielding increasingly lower reductions in high ozone concentrations.

3.5.1: Efficient Regulation of Heterogeneous Pollutants

3.5.1.1: Theory

In general, efficient regulatory instruments set the marginal cost of control equal to the marginal damage caused by those emissions (Hahn & Stavins, 1991; Tietenberg, 1995). In the case of a homogenous pollutant, marginal damages are equal across sources, so the location and timing of emissions are irrelevant to the selected regulatory instrument (Tietenberg, 1990). But when pollutants have heterogeneous impacts, as in the case of NOx emissions, location and timing matter to regulation (Tietenberg, 2010), and differentiated policies are welfare dominating under perfect certainty and complete information (Fowlie & Muller, 2013).

Of course, in the real world, perfect certainty and complete information can never be obtained, especially by regulators, raising the possibility that differentiating regulations could reduce their efficiency (Fowlie & Muller, 2013). For instance, implementing differentiated regulations requires greater data collection and analysis, which imposes a heavy burden on modelers and policy makers and implementers and may incur nontrivial implementation costs (Tietenberg, 1995). The more differentiated the program, the heavier the burden and higher the costs. Fortunately, simpler differentiated trading programs may be just as good as more differentiated ones. (Muller, 2011), for instance, finds that damages from emissions of many electricity generators are statistically equivalent, so in the context of a differentiated cap-and-trade program, groups of generators could be formed within which trading ratios could be set to unity.

Two main differentiated market-based instruments are differentiated taxes and permit markets (Fowlie & Muller, 2013). The former instrument is relatively straightforward: the tax rate is set equal to the marginal damage caused by those emissions (Fowlie & Muller, 2013; Muller & Mendelsohn, 2009; Tietenberg, 1995). Thus, in the case of a spatially- and temporally-differentiated pollutant, the optimal tax rate would vary for each ton of pollutant emitted depending on where and when it is emitted.

The latter approach, a differentiated permit market, is implemented by constraining trading in some fashion. For instance, trading can be prohibited between areas or can be restricted in such a manner that prevents violation of a given standard, e.g. a NAAQS, or any air quality deterioration (Tietenberg, 1995). Of more relevance to this research, differentiated trading ratios could also be implemented between sources, such that permits do not trade on a 1-to-1 basis, but rather on a basis that reflects the damages from the bought and sold emissions (Tietenberg, 1995). In an idealized scenario, trading ratios would be inversely proportional to the marginal damages of the traded pollutants (Muller & Mendelsohn, 2009).

Implementing such a system to account for spatial differentiation in impacts of NOx emissions would require assigning trading ratios between each pair of sources based on their location. But implementing a similar system to account for temporal differentiation is more difficult, since trading may not occur in real-time with emissions. Rather than adjusting trading ratios between sources, though, the ratio of permits that must be surrendered, or used, for each ton of emissions could be changed over time. Thus, on days when NOx emissions are more damaging (such as on high ozone days), more permits would need to be surrendered for each unit of (NOx) emissions.

3.5.1.2: Empirical Studies

Empirical studies, including on NOx emissions, largely corroborate theoretical predictions that differentiating regulation of heterogeneous pollutants will yield efficiency gains. (Muller & Mendelsohn, 2009), for instance, formulate a differentiated SO_2 trading system by setting trading ratios to be inversely proportional to the marginal damage of emissions at each plant. This differentiated trading system is then compared to the Acid Rain Program's undifferentiated trading framework for SO_2. The authors find that the differentiated program results in savings of hundreds of millions of dollars, roughly comparable to the savings attained by moving from "command-and-control" standards to cap-and-trade in the first place. One key to this outcome is that the authors find damages from SO_2 emissions are highly variable across sources; the trading ratio between generators in the 1^{st} and 99.9^{th} percentiles of damages is 49.4:1, for instance, whereas the ratio is 4.3:1 for the 50^{th} to 99^{th} percentile.

While some studies do not find clear efficiency gains from differentiation, these studies are not relevant here. (Krupnick et al., 2000), for instance, compare the efficiency of a differentiated cap-and-trade program for reducing NOx emissions in the eastern U.S. to an undifferentiated one. In the differentiated trading program, plants are grouped into six regions between which permit trading is differentiated based on ozone exposure impacts. The differentiated trading program is found to provide no clear benefit to an undifferentiated one. But this is because variations in pollution effects between the six regions are relatively small, such that trading ratios between most regions are near unity so differentiation yields no clear benefits. Conversely, abatement costs differ between regions, leading to overall greater costs under the differentiated program (on the order of 3%). Yet, there is strong variability in the impacts of NOx emissions across time, particularly on high ozone days. Thus, the underlying factor that undercuts efficiency gains from differentiation in this study does not pertain to differentiating NOx emissions on high ozone days.

3.5.2: Declining High Ozone Concentration Reductions from NOx Emission Reductions under Current Undifferentiated Regulations

A second piece of evidence that suggests policy differentiation could improve the efficiency of reducing peak ozone concentrations is that ozone concentration decreases have lagged NOx emission reductions over the past decade in the Eastern U.S. and Texas. In 2002, (Mauzerall et al., 2005) analyzed NOx and ozone concentrations before and after implementation of the NOx Budget Program (NBP), which reduced NOx emissions from large point sources (mainly power plants) in the Eastern U.S. Despite a decrease of roughly 50% from 1990 levels between 1999 and 2002, Mauzerall et al. did not find a similar decrease in concentrations of NOx (specifically NO) or ozone during the summer season before and after the 1999 NBP cap went into effect.

Recent data on peak ozone concentrations from Texas similarly show declining returns from NOx emission reductions. Data was collected from the Texas Commission on Environmental Quality (TCEQ) on compliance with the 8-hour ozone standard, i.e. the three-year average of the annual fourth highest daily maximum 8-hour ozone concentration measured. Ozone concentrations were gathered from monitoring stations across the state, including in all major metropolitan areas. The three-year average values provide a fair index for ozone concentrations on high-ozone days. Total and power sector NOx emissions in Texas were obtained from the National Emissions Inventory, specifically for years 2002, 2008 and 2011 (U.S. Environmental Protection Agency, 2011d). As such, ozone data was obtained for those same years.

Between 2008 and 2011, total and power sector NOx emissions in Texas fell by roughly 14% and 7%, respectively (U.S. Environmental Protection Agency, 2011d). During that same period, the state average 8-hour ozone standard value declined by just 1% (Texas Commission on Environmental Quality, 2014a). Yet, between 2002 and 2008, an 8% and 42% reduction in total and power sector NOx emissions, respectively, yielded a 10% reduction in the state average 8-hour ozone standard value. In other words, the same NOx emission reduction returned a 40% lower reduction in the state average 8-hour ozone standard concentration between 2008 and 2011 than between 2002 and 2008. This data must be viewed with considerable caution, given that many other factors, including meteorology, influence ozone formation and NOx emissions are emitted by many more sectors than just power plants. Even so, the data further indicate that the current regulatory approach to achieving compliance with the ozone NAAQS, i.e. to reducing peak ozone concentrations, is becoming less effective.

This declining efficacy may in part be due to the behavior of power plant operators under undifferentiated cap-and-trade programs. Analysis by (Martin, 2008) suggests that some power plant operators reduce power output, and consequently NOx emissions, early in the summer, allowing them to emit more NOx later in the summer while remaining under their summer-wide NOx emissions cap. As Martin points out, this behavior makes perfect sense economically. In the United States, power demand typically peaks on hot sunny days due to air conditioning loads. To meet this demand, many power plants on the grid have to be operated. Furthermore, when electricity demand peaks, so too do prices, creating a large financial incentive for power plants to generate power on those days. Consequently, power plants make the most money on these days, and so save emission permits to ensure they are able to operate on these days.

Yet, it is on the same hot sunny days when demand peaks that meteorological conditions favor greatest damages from NOx emissions through high rates of ozone formation (Mauzerall et al., 2005; U.S. Environmental Protection Agency, 2011a). Consequently, the deferral of emissions to high demand days undermines the ability of the current undifferentiated cap-and-trade program to reduce emissions on high ozone days and therefore formation of ozone on those days.

3.6: Summary

Over the past decade, temporally-undifferentiated cap-and-trade programs like the extant CAIR have been used to achieve compliance with the ozone NAAQS by reducing NOx emissions. These programs treat NOx emissions in the summer uniformly. Yet, NOx emissions have highly variable damages depending on when they are emitted within the summer (Mauzerall et al., 2005). Specifically, NOx emissions have much greater damages on hot sunny days when ozone formation is greatest. Consequently, reducing NOx emissions that contribute more to high ozone days would also yield reductions in more harmful NOx emissions – a strong public health motivation for reducing NOx emissions that contribute to high ozone concentrations. Moreover, states are legally required to reduce ozone concentrations in excess of the NAAQS under the Clean Air Act, which is typically achieved through NOx emission reductions.

Thus, strong public health and legal rationales exist for reducing NOx emissions that contribute to high ozone concentrations. The question then becomes what regulatory instrument should be used to achieve these reductions. Two lines of evidence indicate a time-differentiated market-based instrument would be the most efficient option. First, economic theory and empirical studies indicate that differentiating regulations of heterogeneous pollutants like NOx improves regulatory efficiency. Second, NOx emission reductions achieved under undifferentiated programs in recent years have yielded declining returns in ozone concentration reductions, suggesting declining efficacy of these programs. In light of these facts, the next chapter proposes a time-differentiated pricing scheme for reducing NOx emissions that contribute to ozone formation on high ozone days.

4.0: Proposed New Regulation for Reducing NOx Emissions from Power Plants on High Ozone Days: Time-Differentiated Pricing

In the previous chapter, the case was made for using a time-differentiated market-based instrument to reduce NOx emissions that contribute more to ozone formation on high ozone days. This chapter proposes a specific time-differentiated regulatory strategy that would cost-effectively reduce NOx emissions from the power sector that have the greatest impact on ozone exceedances. A necessary simplifying assumption is made that NOx emissions that occur on high ozone days are the greatest contributors to ozone formation on high ozone days. This may or may not be true, but determining its veracity requires detailed air quality modeling that is beyond the scope of this work. Thus, the time-differentiated pricing instrument presented here aims to reduce NOx emissions on high ozone days.

The chapter begins by describing the proposed time-differentiated pricing system. It then discusses how the system would alter short- and long-term behavior of power plants. The two prior studies on the type of time-differentiated pricing proposed here are subsequently discussed. The chapter concludes with the novel contributions of this thesis.

4.1: Description of Time-Differentiated Pricing

Time-differentiated pricing is a differentiated regulatory instrument that has been previously proposed for efficiently reducing NOx emissions from power plants on high ozone days (Bharvirkar et al., 2004; Sun et al., 2012). In general, time-differentiated pricing aims to reduce emissions of a pollutant when its impacts are greatest by temporarily assessing a high price on emitting it. Here, time-differentiated pricing is applied to NOx emissions on high ozone days from power plants; on forecasted high ozone days an elevated price per ton of NOx emissions would be assessed for each generator from midnight to midnight.

Given that the lifetime of NOx particles near the surface is often less than 24 hours (U.S. Environmental Protection Agency, 2006), although can be as long as a few days (Bharvirkar et al., 2004), a 24-hour price window may significantly decrease ambient NOx concentrations on high ozone days. Depending on local meteorology and VOC concentrations, lower NOx concentrations may in turn lead to less ozone formation, mitigating ozone concentrations on the worst days.

The time-differentiated pricing regime proposed here is not “fully” differentiated, in that prices do not vary spatially, but only vary temporally at a 24-hour interval. Under imperfect information and uncertain outcomes, though, such a simpler differentiated program may be as good as or even better than a “fully” differentiated program (Fowlie & Muller, 2013; Muller, 2011). Simpler programs are also less costly and onerous for states and administrators (Tietenberg, 1995), making them more amenable to implementation.

For the remainder of this thesis, “time-differentiated pricing” will be used as shorthand, unless specifically indicated as being used in the abstract, to refer to this conception of a time-differentiated price; i.e., one that sets a higher price on NOx emissions on high ozone days.

4.2: Effect of Time-Differentiated Pricing on Power Plant Operations

Time-differentiated pricing can reduce NOx emissions through short- and long-term effects on power plant operations (Bharvirkar et al., 2004; Sun et al., 2012). In the short-term, power plants incorporate the emissions price into their cost of power generation on days when the price is triggered, leading to reduced power generation at plants with greater NOx emissions rates. In the long-term, power plants can mitigate the added cost of time-differentiated pricing by installing control technologies to reduce their NOx emissions rate. In this research, both short- and long-term effects of time-differentiated pricing will be accounted for.

4.2.1: Short-Term Response to Time-Differentiated Pricing: Redispatching

In the short-term, power plants would internalize a time-differentiated price on NOx emissions into their operating cost. To understand what effect this would have, a brief primer on electricity markets is first given.

The two power systems studied here, in the Mid-Atlantic and Texas, have competitive electricity markets operated by Independent System Operators (ISOs), specifically the Pennsylvania-New Jersey-Maryland Interconnect (PJM) and the Electric Reliability Council of Texas (ERCOT), respectively. In competitive electricity markets, power plants compete to generate power by submitting bids to the ISO with how much power they will generate at what price. The ISO then takes those bids and runs a dispatch model (specifically, a security-constrained unit commitment) that meets demand with the given bids while minimizing total system costs. With the model's output, the ISO then dispatches generators, meaning the ISO tells each power plant how much power it can generate, in order to meet demand at least cost. As a result, setting transmission constraints aside, cheaper power plants are dispatched by the ISO first (Batlle, 2013).

In this context, the short-term effect of a time-differentiated price can be understood. When the price is triggered, a power plant will include its cost of emissions under the price in its operating cost, which it then submits to the ISO. Power plants with greater NOx emissions rates will have higher operating cost increases under the time-differentiated price. Since the ISO will generally dispatch less costly generators first, power plants with higher NOx emissions rates – and consequently higher operating cost increases – will be dispatched less. Conversely, power plants with lower NOx emissions rates will be dispatched first and generator more power. Total NOx emissions when the price is triggered will therefore decrease as more power generated from less NOx-intensive power plants.

4.2.2: Long-Term Response to Time-Differentiated Pricing: Control Technology Installations

In the long-term, power plants can install control technology in order to mitigate the effect of a time-differentiated price on their operating cost. Control technologies reduce emissions rates of power plants through various means. In the case of NOx, several commercially-viable combustion and post-combustion control technologies exist to reduce NOx emissions (EPA 2010). Combustion control technologies, like low-NOx burners, reduce NOx emissions by inhibiting its formation during fuel combustion by regulating temperature and flame conditions (U.S. Environmental Protection Agency Office of Air and Radiation, 2010). Most plants have already installed NOx combustion control technologies (U.S. Environmental Protection Agency, 2012b) in order to comply with prescriptive emissions rate standards (Burtraw & Evans, 2003). With regard to post-combustion control technologies, two exist for NOx, Selective Catalytic Reduction (SCR) and Selective Non-Catalytic Reduction (SNCR). Given already widespread adoption of combustion controls, this research focuses on post-combustion controls. More specifically, this research focuses on SCR because it offers greater potential for emission reductions and is more common than SNCR, but still not widely adopted.

According to the EPA, SCR can reduce NOx emissions rates of coal-fired generators by up to 90% (U.S. Environmental Protection Agency Office of Air and Radiation, 2010). Installing and operating SCR on high ozone days would therefore mitigate most of the increase in operating cost of a time-differentiated price. With that lower operating cost, the plant may be dispatched when it otherwise would not have been and generate significant profits, given that high ozone days tend to have high electricity prices due to high power demand. Thus, although SCR capital costs are quite high, approximately $90 million for a 500 MW coal plant (U.S. Environmental Protection Agency Office of Air and Radiation, 2010), large profits reaped on high ozone days may compensate for the high capital costs of SCR.

4.3: Prior Studies on Time-Differentiated Pricing

Because time-differentiated pricing is a relatively-new potential tool for reducing high-ozone days, only two studies have assessed its merits. (Bharvirkar et al., 2004) consider NOx emission reductions from time-differentiated pricing on high-ozone days in Maryland, part of the PJM Interconnect. To do so, they used the Haiku model, which uses a simple economic dispatch algorithm to determine power generation at power plants (Paul, Burtraw, & Palmer, 2009). Economic dispatch models ignore real-world operational constraints, like ramping limits and minimum load, when dispatching generators, instead relying solely on the variable cost of each plant and its maximum capacity. The Haiku model captures control technology installations, in this case Selective Catalytic and Non-Catalytic Reduction (SCR and SNCR), in response to policies. Because the Haiku model aggregates demand into time blocks, high-ozone episodes are not modeled on a daily basis, but rather the entire month of July is considered a high-ozone episode.

Bharvirkar et al. found that time-differentiated pricing could significantly reduce NOx emissions. The reduction was largely driven by control technology installations at coal and gas plants; across various time-differentiated prices, all coal plants opted to install some form of control technology, as did most gas plants. Fuel substitution of gas for coal also contributed to reduced NOx emissions, though. Alternative control scenarios were not tested in this study.

(Sun et al., 2012) provided a more recent and relevant analysis of the effect of time-differentiated pricing on NOx emissions and ozone concentrations in PJM Classic, which includes Pennsylvania, Maryland, Delaware, and New Jersey. Power plants were dispatched using an optimal power flow (OPF) model. OPF models dispatch generators with an economic dispatch algorithm while accounting for transmission constraints. Plants were not given the option of installing control technologies, but alternative technology-based standards were modeled that mandate SNCR or SCR adoption on all or 32% of coal plants, respectively. The SCR scenario was calibrated such that the same NOx emission reductions were achieved as in one of the time-differentiated pricing scenarios, $100,000 per ton NOx emitted.

Sun et al. demonstrated that sufficient flexibility on the grid existed to meet demand, even during peak demand hours, under time-differentiated pricing. Moreover, significant NOx emission reductions of 20% or more were achieved on high-ozone days. Declining NOx emissions, in turn, reduced ozone concentrations by 1 ppb or more in large portions of major metropolitan areas like Philadelphia. Reductions in NOx emissions from time-differentiated pricing were largely comparable to those achieved under the SNCR and SCR scenarios.

Regarding cost-effectiveness for reducing NOx emissions on high ozone days, time-differentiated pricing was compared to the SCR and SNCR scenarios at various high-ozone thresholds and ozone forecast accuracies. High-ozone thresholds were based on observed atmospheric ozone concentrations. In general, across many of the thresholds and forecasting accuracies, time-differentiated pricing was more cost-effective. At a low (strict) threshold for determining high-ozone days, time-differentiated pricing was more cost-effective than the SNCR, but not SCR, scenario. At a high high-ozone threshold, time-differentiated pricing was more cost-effective than both SCR and SNCR scenarios. These findings were robust to ozone forecast uncertainty; under current ozone forecasting error rates, time-differentiated pricing was often more cost-competitive than the SCR and SNCR scenarios modeled.

These two studies, taken together, suggest that time-differentiated pricing can significantly reduce NOx emissions on high ozone days, which in turn may reduce ozone concentrations on high-ozone days. Furthermore, time-differentiated pricing may provide a cost-effective means of reducing NOx emissions relative to alternative technology-based standards.

However, both studies suffer from several methodological shortcomings. First, neither study uses a sophisticated dispatching algorithm that accounts for real-world operational constraints on generators, such as minimum load and ramping constraints. These constraints are particularly important given that time-differentiated pricing would be triggered sporadically throughout the summer, meaning generators would have to continually shift from normal operations to accommodate the pricing scheme. Second, while (Bharvirkar et al., 2004) model control technology installation decisions at each plant, these decisions are made based on an overly-simplified power system model that is ill-suited to fully capturing power plant operations of a time-differentiated pricing regime, as discussed above. (Sun et al., 2012), on the other hand, do not allow for control technology installations at all. Third, neither study compares time-differentiated pricing to alternative regulations. The most comparative analysis that is done is in (Sun et al., 2012), where time-differentiated pricing is compared to just two technology-based standards. Fourth and finally, both studies are done in the same power system, PJM, which is dominated by large inflexible coal plants. But the effects on costs and emissions of time-differentiated pricing may vary depending on the fuel mix of a power system. For instance, in systems with high penetrations of flexible natural gas units, time-differentiated pricing may lead to greater fuel substitution than found by (Bharvirkar et al., 2004).

4.4: Purpose and Novel Contributions of Thesis

The question of which regulatory instrument is the most efficient for reducing high ozone concentrations is a highly complex one that spans multiple disciplines. This thesis informs a key component within that question: the cost-effectiveness of various regulatory instruments at reducing NOx emissions that may contribute to ozone formation on high ozone days. Indeed, information generated in the research and presented here is necessary to determining the efficiency of the assessed policies, since emissions must be used in an air quality model to calculate air quality changes and consequent public health benefits. These benefits could then be monetized and compared to costs calculated here to determine the efficiency of each policy. In fact, this research is part of a larger undertaking with a team at the University of Texas in Austin led by Dr. Elena McDonald-Buller to carry out air quality modeling on emissions generated here.

This thesis contributes substantially to the existing literature on the merits of a time-differentiated pricing scheme for reducing NOx emissions on high ozone days. For the first time, time-differentiated NOx pricing is compared to undifferentiated pricing and technology-based standards, the latter of which is modeled via Monte Carlo analysis. Modeling technology-based standards in this manner is a new attempt to capture the real-world unpredictability in the crafting and implementation of such standards.

This thesis also provides the most detailed picture to date of the impact of time-differentiated pricing and other policies on system-wide emissions and costs through a novel two-phase model. The two-phase model captures the short- and long-term response of generators to time-differentiated pricing and other policies. Embedded in the two-phase model is a power system model, called a unit commitment model, that is more complex than those used in past studies on time-differentiated pricing (Bharvirkar et al., 2004; Sun et al., 2012). This more complex power system model accounts for operational characteristics of generators that can significantly affect the impact of policies on emissions and costs. The two-phase model developed here also adopts a game theoretic approach to determining control technology installations in response to policies. Nash equilibrium is determined for control technology installations among generators, thereby accounting for the effect control technology installation at one generator can have on all other generators through electricity prices. The application of Nash equilibrium here is the first in this context.

Chapter 5: Methodology

The models and data sources used in this research are detailed below. Short- and long-term responses of electricity generators to policies are captured with a two-phase model. In the first phase, control technology installations are determined on the basis of profit maximization. In the second phase, emissions and cost are calculated under each policy given control technology installations from the first phase. Within each phase, a detailed unit commitment model is run to determine hourly power generation from each electricity generator. Based on the output from this model, profitability with and without control technologies in phase one and system-wide emissions and costs in phase two are determined.

5.1: Study Systems

Two study systems are used in this research: the Electric Reliability Council of Texas (ERCOT) and a part of the Pennsylvania-New Jersey-Maryland Interconnect (PJM) called PJM Classic. While ERCOT is an Independent System Operator (ISO) and PJM is a Regional Transmission Operator (RTO), both serve essentially the same function in areas with competitive electricity markets: they operate the transmission grid, administer the electricity wholesale market, and dispatch generators, i.e. dictate their power output, to meet demand at least cost.

ERCOT operates nearly the entire grid in Texas except for the panhandle and other scattered locations. It encompasses roughly 85% of the electricity supply and demand in Texas, and has a very weak interconnection (roughly 2 gigawatts (GW)) with the surrounding power systems, making it an essentially isolated power system. Peak demand in 2012 in ERCOT was roughly 66.5 GW, set in June (D&E report). ERCOT's grid is predominantly natural gas – in 2012, roughly 57% of its generation capacity was natural gas, followed by coal at 23%, wind at 13%, and nuclear at 6% (Dumas, 2013).

PJM's control area includes all of Pennsylvania, New Jersey, Delaware, Maryland, Virginia, West Virginia, Washington D.C., and Ohio, as well as parts of Kentucky, Tennessee, North Carolina, Illinois (including Chicago), Michigan, and Indiana. System peak in PJM in 2012 was 154 GW (PJM Resource Adequacy Planning Department, 2013). PJM's capacity is roughly 41% coal, 19% nuclear, 16% natural gas, 8% oil, and the remainder hydro, renewables and other gas (PJM, 2012). PJM is part of the Eastern Interconnection, one of the three grids in the United States that includes about the eastern half of the nation. As a result, large amounts of power flow into and out of PJM to neighboring power systems.

The complete PJM system is too large to model with a traditional unit commitment model, the primary tool used in this research as discussed below. Consequently, PJM Classic is used, which includes Pennsylvania, New Jersey, Maryland, Delaware and Washington, D.C. In PJM data and planning documents, PJM Classic may be referred to as PJM Mid-Atlantic or PJM East. The mix in PJM Classic is 33% coal, 30% gas, 20% nuclear, 9% oil, 4% hydro, 2% wind, and the remainder predominantly biomass and solar (U.S. Energy Information Administration, 2013a).

5.2: Two-Phase Model for Determining Selective Catalytic Reduction Installations, Emissions and Costs

A two-phase model was constructed to determine system-wide costs and emissions under differentiated pricing and other policies. In the first phase, the model determines which plants would install control technology, specifically Selective Catalytic Reduction, under the given pricing scheme or policy based on profit maximization. In the second phase, given those decisions, costs and emissions are calculated from power generation over the entire summer. In both phases, a unit commitment model is used to dispatch, i.e. determine hourly power generation at, electricity generators. The unit commitment model formulation is presented first, followed by an elaboration on each phase of the two-phase model developed for this work.

5.2.1: Unit Commitment Model

5.2.1.1: Formulation

5.2.1.1.1: Unit Commitment Model Overview

A unit commitment (UC) model is a mixed integer linear program that optimally dispatches power plants to meet demand at least cost. UC models account for various real-world operational constraints on power plants, such as limits on ramping (the rate of power output increase or decrease) and minimum uptime or downtime (the minimum time between unit shut-down and start-up). Input to the model includes dispatchable power plants, plant parameters including power capacity and operational cost, and demand. The model results include hourly power generation at each generator as well as the hourly cost of electricity, extracted as the marginal cost on balancing supply and demand. The model is implemented in the General Algebraic Modeling System (GAMS) Version 24.1 (GAMS Development Corporation, 2013), and is solved using CPLEX Version 12.

5.2.1.1.2: Treatment of the Transmission Network

Unit commitment models typically do not include the transmission network for two reasons. First and foremost, the transmission grids in U.S. power systems are well-meshed, such that significant transmission constraints rarely occur, especially during off-peak hours. Thus, including the transmission network would not drastically change model output. Second, there is a practical modeling limitation in that including the entire transmission network would result in a prohibitively-large optimization problem.

Given the above and the fact that PJM operates a well-connected transmission network, the PJM Classic model used here does not consider the transmission network. Consequently, all electricity demand and generation is essentially located at a single node. ERCOT differs from PJM in that the entire system is not fully meshed. Rather, the transmission network in ERCOT's territory is zonally constrained. In other words, within zones the transmission network is well-meshed and constraints are not significant, but between zones there are significant transmission constraints that limit power flow between zones (Baldick & Baughman, 2003). So, here ERCOT is modeled as a zonally-constrained system with four zones: North, Houston, South and West (Baldick & Baughman, 2003). Generators and demand are divided into the appropriate zone, and power flow between zones is constrained per historic data. Specifically, power flow is only permitted between the North and each other zone per 2010 transmission constraints, the most recent year for which data is available (ERCOT 2010). Transmission flows across system boundaries in both ERCOT and PJM are not included in the UC model used here.

5.2.1.1.3: Treatment of Non-Served Energy

UC models often include a non-served energy (NSE) variable, which accounts for periods of excess demand that supply cannot meet (Morales-españa et al., 2013). In reality, though, NSE may be included in UC solutions even when sufficient capacity exists to meet demand in all hours due to the nature of the optimization algorithm, which does not necessarily find the true optimal solution. Furthermore, it is not at all clear that there is any unmet electricity demand in the United States due to excess prices. Indeed, in ERCOT the price cap, or the maximum price that electricity is allowed to reach, has been hit before without any load going unserved (Acclaim Energy Advisors, 2014). Unmet demand certainly occurs during blackouts, planned or otherwise, but such events occur during extreme events (Reuters, 2014) and are fundamentally different than having demand go unmet in a model that is operating under normal dispatch circumstances.

The cost of NSE is set very high in UC models, as it represents the cost to society of having power demand go unmet. ERCOT and PJM both set offer caps on electricity, which are used here as the cost of non-served energy, as no generators would produce electricity above this price. In 2012, the offer cap was $4,500 per megawatt-hour (MWh) in ERCOT (Texas Administrative Code § 25.808) and $1,000 per MWh in PJM (Monitoring Analytics, 2013).

As discussed at length later, the first phase of the model determines control technology installations via profit maximization. For each plant, profits with and without a control technology are simulated, and if profits are greater with the control technology than without, the control technology is installed. This calculation is heavily distorted by the presence of any NSE in solutions because of the extremely high cost of NSE, which sets the marginal cost of electricity in any hour it is present. If a plant were to operate in an hour with NSE with but not without a control technology, then that hour of operations alone would generate so much profit as to make it seem profitable to install control technology. But as stated previously, such instances of NSE occur very rarely, if ever, in the United States. Thus, their inclusion in UC results represent an artifact more than a realistic representation of system changes when a control technology is installed.

In light of these facts, this research utilizes a UC formulation with and without NSE. The UC model without NSE is used in the first phase of the model, when control technology installation decisions are made. The UC model with NSE is used in the second phase, when emissions and costs are determined for meeting summer demand. Because removal of NSE makes finding an optimal solution more difficult, a wider optimality gap of 1E-3 is used than the 1E-4 used in the latter formulation. The optimality gap essentially determines the quality of a solution the integer solver must get before quitting. More specifically, it is the allowable gap between the "best possible" integer solution and the solver's current "best estimate" of the solution.

5.2.1.1.4: Detailed Unit Commitment Model Formulation

The UC formulation used here is based on the formulation in (Morales-españa et al., 2013). The full UC formulation in GAMS is given below for ERCOT including non-served energy (NSE), although an alternate version of the ERCOT model excludes NSE. For PJM, two model formulations are used with and without NSE. Since PJM is modeled as a single node, the PJM model does not include any zones or transmission lines.

Table 2: Terms in UC model as implemented in GAMS for the ERCOT power system with NSE. Terms associated with transmission constraints and non-served energy are not included in alternate UC model formulations.

Sets	
t	hour of week, $t \in T$
i	generator, $i \in N$
z	ERCOT demand zone (North, Houston, South or West), $z \in Z$
l	transmission lines between zones (to and from North and each other zone), $l \in L$
Parameters	
$P_{z,t}^{D}$	demand in hour t in zone z [GW]
P_t^{R}	system spinning reserves in hour t, equal to 1% of system demand in hour t [GW]
$F_{l,t}^{MAX}$	maximum flow over transmission line l in hour t [GW]
RL_i	ramping limit (up and down) for unit i [GW]
P_i^{MIN}	minimum power output of unit i [GW]
P_i^{MAX}	maximum power output (capacity) of unit i [GW]
SU_i^{FIXED}	start-up fixed cost for unit i [thousands $]
SU_i^{FUEL}	start-up fuel cost for unit i [thousands $]
MDT_i	minimum down time for unit i [hours]
MUT_i	minimum uptime for unit i [hours]
$O_{i,t}$	operating cost for unit i in hour t [thousands $/GWh]
LZ_l^{IN}	zones that line l transmits power into {1,2,3,4}
LZ_l^{OUT}	zones that line l transmits power out of {1,2,3,4}
Scalars	
$CNSE$	cost of non-served energy [thousands $/GWh]
Variables	
$nse_{z,t}$	non-served energy in zone z at hour t [GW]
$p_{i,t}$	total power output of unit i in hour t [GW]
$g_{i,t}$	power output above minimum load of unit i in hour t [GW]
$u_{i,t}$	binary variable indicating unit i is operating above its minimum load in period t {0,1}
$w_{i,t}$	binary variable indicating unit i shuts down in period t {0,1}
$v_{i,t}$	binary variable indicating unit i starts up in period t {0,1}
$f_{l,t}$	line flow over transmission line l in hour t [GW]

The UC model minimizes total system costs, the objective function. Total system costs are calculated as the sum of operational and start-up costs. In some formulations, the cost of NSE is also included in total system costs.

$$TC = \sum_{i,t}[p_{i,t} * O_{i,t} + v_{i,t} * (SU_i^{FIXED} + SU_i^{FUEL})] + \sum_{z,t} nse_{z,t} * CNSE$$

Start-up costs include terms for fixed and fuel costs, the latter of which includes emissions costs from initial fuel consumption if applicable. Fixed and fuel start-up costs were based on each generator's size and prime mover type. Operational costs include variable operation and maintenance (O&M), fuel and emissions costs. Generator-specific fuel and emission costs per megawatt of power generated were calculated using generic or, where available, generator-specific heat rates. Constant heat rates are used for all thermal power plants.

The UC model balances hourly supply and demand. In the ERCOT formulation, supply and demand are balanced for each zone with each zone's generators and power flows into and out of each zone. NSE for each zone may also be included. In PJM, demand and generators are located in a single node, so supply and demand are balanced system-wide and no power flows are included.

$$P_{z,t}^{D} = \sum_{i} p_{i,t} + nse_{z,t} + \sum_{l \in (LZ_l^{IN} = z)} f_{l,t} - \sum_{l \in (LZ_l^{OUT} = z)} f_{l,t}$$

Transmission constraints between zones in the ERCOT formulation are constrained to match historic observations.

$$f_{l,t} \leq F_{l,t}^{MAX}$$

In both ERCOT and PJM, the model enforces system-wide (not zonal) hourly spinning reserves at a level of 1% of hourly demand. Spinning reserves refers to spare power generation capacity in generators already producing power. Spinning reserves are therefore calculated as the difference between the maximum output and current power output of all plants.

$$P_t^R \leq \sum_i [u_i^t * P_i^{MAX} - p_i^t]$$

Generators are marked as either on or off through three binary variables. Here, when generators are on, they are generating power above their minimum output. One binary variable (*u*) indicates whether the generator is on or off in each hour, while the other two indicate whether the plant turns on (*v*) or off (*w*) in each hour.

$$u_{i,t} = u_{i,t-1} + v_{i,t} - w_{i,t} \qquad \text{for all } t \geq 2$$

Generator start-ups and shut-downs are constrained by minimum up and down times. Generators cannot start-up after shutting-down until its minimum down time has elapsed. Similarly, a generator cannot shut-down after starting-up until its minimum up time has elapsed.

$$1 - u_{i,t} \geq \sum_{i > t - MDT}^{t} w_{i,t} \qquad \text{for all } t \in [MDT, T]$$

$$u_{i,t} \geq \sum_{i > t - MUT}^{t} v_{i,t} \qquad \text{for all } t \in [MUT, T]$$

Generator power output is represented by two variables in the model: total power output, $p_{i,t}$, and its output above its minimum load, $g_{i,t}$. The maximum total power output of generators is constrained to maximum capacity.

$$p_{i,t} \leq P_i^{MAX}$$

Generation above minimum load is then calculated based on each unit's total power output and minimum power output, P^{MIN}.

$$g_{i,t} = p_{i,t} - u_{i,t} * P_i^{MIN}$$

Generation above minimum output is constrained between the plant's maximum and minimum power outputs.

$$g_{i,t} \leq u_{i,t} * (P_i^{MAX} - P_i^{MIN})$$

Ramping limits, which are the same for up and down ramps, are enforced on each unit's generation above its minimum load.

$$-RL \leq g_{i,t} - g_{i,t-1} \leq RL \qquad \text{for all } t \geq 2$$

5.2.1.2: Data Sources for Unit Commitment Models

5.2.1.2.1: ERCOT

Zonal power demand data for ERCOT is available only up to 2010. As such, 2012 system-wide demand (ERCOT, 2013a) was used, but was divided into zones according to the proportion of demand in each zone in 2010 (ERCOT, 2011). A list of operating generators in ERCOT in 2012 was obtained from two sources.

Generators in ERCOT as of 2009 were obtained from the Emissions and Generation Resource Integrated Database (eGRID) (U.S. Environmental Protection Agency, 2012c). Plants in eGRID that were marked as retired, out of service, or not in any of the four ERCOT zones were discarded. The generator fleet was then updated to 2012 using ERCOT's Capacity, Demand and Reserves (CDR) reports for 2009 and 2012 (ERCOT, 2012a), which list plants that are in service, retired, mothballed, or enter service each year. In total, 607 generators are included in our 2012 model.

Generic fuel costs for the electric power sector in 2012 were obtained from the EIA (U.S. Energy Information Administration, 2013b). All costs used throughout this work were discounted to 2007 dollars using Consumer Price Index data (U.S. Bureau of Labor and Statistics, 2013). When possible, generator-specific average emissions rates were calculated from Continuous Emissions Monitoring Systems (CEMS) data (U.S. Environmental Protection Agency, 2012a) for October 2011 to September 2012. CEMS data provides hourly power output, emissions and fuel input data for facilities covered under the Acid Rain Program, which includes nearly all of the coal facilities in ERCOT. When CEMS data was not available, generator heat and emissions rates were set equal to eGRID plant-level data for eGRID generators. For generators from ERCOT's CDR reports, generator heat and emissions rates were set equal to average values per fuel type calculated from CEMS or, for biomass facilities, eGRID data. Emissions rates for natural gas combined cycle steam turbines were set equal to the plant-level average to prevent CCSTs from being dispatched when the plant's combustion turbines were not generating power.

Generator capacities were modified to reflect real-world operations. Combined heat and power (CHP) generator capacities were determined with data on electricity sales to the grid from Form 923 of the Energy Information Administration (U.S. Energy Information Administration, 2012). Form 923 collects annual data on the fate of electricity generated at power plants, e.g. whether the electricity is sold to the grid or used on-site. CHP generators were either excluded from analysis, or their capacity was set to half or all of their nameplate capacity per eGRID (see

Appendix A for further discussion). The maximum output of gas, coal and nuclear plants was limited to 95% of nameplate capacity to better represent real-world operations. Hourly wind and solar output data for 2012 were obtained from ERCOT (ERCOT, 2012b) and, in conjunction with the total installed capacity of wind and solar, used to calculate hourly wind and solar capacity factors. The hourly capacity of each wind and solar generator was scaled down by the appropriate hourly capacity factor prior to inclusion in our unit commitment model as dispatchable.

5.2.1.2.2: PJM

Hourly demand data for 2012 provided by PJM (PJM Interconnect, 2014a) is given at the system-level as well as in smaller divisions. Based on a map of transmission zones within PJM (PJM Interconnect, 2014b), PJM Classic states (PA, NJ, MD, DE and DC) were mapped to transmission zones (PS, PE, PL, BC, GPU, PEP, CNCT and RECO). These transmission zones are collectively labeled as PJM Mid-Atlantic by PJM, so hourly demand data given by PJM for this region was used.

Generators operating in 2012 in PJM were obtained from the EIA's 2012 Form 860 (U.S. Energy Information Administration, 2013a). Form 860 collects information on electricity generating units annually. It provides information at the plant and generator level, which were merged for this research. Plants in PJM were only included if they were located in the PJM Classic states. Generators that were marked as retired or out of service were not used in this research, nor were generators that did not deliver power to the transmission grid as indicated in Form 860.

As in ERCOT, generic fuel costs for the electric power sector were obtained from the EIA (U.S. Energy Information Administration, 2013b). Emissions rates provided in Form 860 were replaced where possible with average emissions rates calculated from 2012 CEMS data (U.S. Environmental Protection Agency, 2012a). Combined cycle steam turbine emissions rates were set equal to the plant-level average, as in ERCOT, to prevent dispatching without combustion turbines.

Generator capacities were also modified in the same ways as in ERCOT. Capacities of CHP units were tailored based on reported data on EIA Form 923 (U.S. Energy Information Administration, 2012). Maximum capacities of nuclear, coal and gas plants were limited to 95% of their nameplate capacity. Wind and solar capacities, of which there is very little in PJM Classic, were set hourly based on historical data. Hourly wind generation in 2012 was obtained from PJM (PJM Interconnect, 2012), whereas 2012 solar data from ERCOT (ERCOT, 2012b) was used because PJM does not provide hourly solar data. While PJM and ERCOT have different hourly insolation, there are negligible levels of solar in PJM, so no distortionary effects are anticipated from this assumption. Wind and solar generation data were used to calculate hourly capacity factors for wind and solar facilities, which were then included in the UC model as dispatchable.

5.2.2: First Phase: Determining Control Technology Installations

The first phase of the two-phase model determines control technology installations at coal-fired generators. Control technology installation decisions are made at the generator, not plant, level. The strategy used in the model is the same for PJM and ERCOT. In general, the model determines installations at coal-fired generators of Selective Catalytic Reduction (SCR), a post-combustion control technology for NOx emissions. Only coal-fired generators that do not already have a post-combustion control installed for NOx emissions are considered. Data on existing SCR and SNCR installations are obtained from EPA's Acid Rain Program facility information (U.S. Environmental Protection Agency, 2012b).

Each coal-fired generator decides whether to install SCR via profit maximization; profits with and without SCR from power generation are calculated for each generator, and if profits are greater with SCR, it is installed. All SCR costs are accounted for in this calculation, including fixed capital and O&M costs and variable operational costs. Variable operational costs of SCR include a variable O&M cost and a heat rate penalty.

Once all installation decisions are made unilaterally, each generator is allowed to reconsider its decision given the decisions of all other generator. This process iterates until Nash equilibrium is reached, meaning no generator changes its decision given the decision of all other generators. Nash equilibrium has never been applied to determining control technology installations with a power system model before. Its application here is crucial to capturing the market dynamics of control technology installations, as installations at other plants affect system-wide prices and therefore the profitability of installations at other plants.

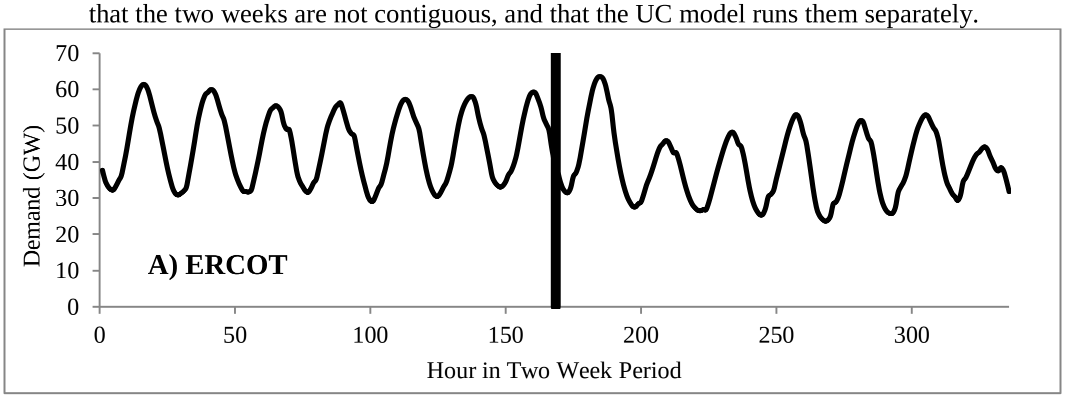
SCR is a post-combustion NOx control technology that significantly reduces NOx emissions, slightly reduces operational efficiency by consuming some of the plant's power output, and incurs additional variable O&M costs. Consequently, SCR installation is simulated as a reduction in NOx emissions, slight decrease in heat rate, and increase in variable O&M using values from EPA's Base Case v4.10 (U.S. Environmental Protection Agency Office of Air and Radiation, 2010). The latter two values depend on the size and heat rate of the plant (see Appendix A for values). Reduction of NOx emissions by SCR is set to 90%, but the lower limit of NOx emissions with SCR is set to 0.06 pounds per MMBTU (U.S. Environmental Protection Agency Office of Air and Radiation, 2010), which constrains the removal efficiency of SCR by many coal plants. SCR operation is considered dispatchable (Patiño-Echeverri et al., 2007), such that coal plants can turn SCR on and off on days when it is and is not economical to run.

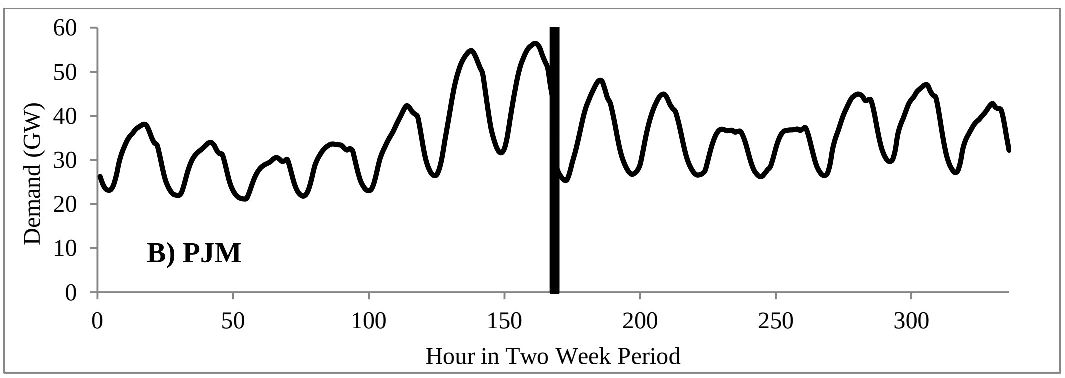
At each coal plant, profits from power generation are calculated with and without SCR. Power generation is determined with the previously-described UC model. Given the computational intensity of our UC model, it was not feasible to simulate operations with and without SCR for each coal plant over the entire summer. Instead, the UC model is run for two non-contiguous summer weeks that best represent the entire summer demand profile in each power system. By using two weeks that are highly representative of summer demand, profits with and without SCR over this two week period roughly approximate profits over the entire summer with and without SCR.

The algorithm to select the two weeks was taken from (Sisternes & Webster, 2013). Specifically, to select the two weeks, summer demand was converted to net load by subtracting generation from renewable resources, and a load duration curve was created from the net load. Then, a summer-long load duration curve was created by expanding each possible combination of two weeks. Those two weeks with the lowest normalized root-mean-square error to the actual net load duration curve were selected for use in the UC model. To calculate net load in ERCOT, hourly wind and solar generation data from ERCOT in 2012 were used (ERCOT, 2012b). In PJM, hourly wind data from PJM in 2012 was used, but solar data is not provided by PJM. As such, solar data from ERCOT was also used here, but very little installed capacity of solar exists in PJM (roughly 370 MW).

For ERCOT, the two weeks used, June 29 to July 5 and September 7 to 13, have a normalized root-mean-square error of 1.23%. For PJM, the NRMSE for the two weeks, June 15 to 21 and August 31 to September 6, is a similarly-low 1.55%. Figure 3 below depicts the demand profile in PJM and ERCOT for the two weeks over which SCR installation is tested.

Figure 3: Demand profile for the two weeks selected as most representative of summer demand and therefore used to determine SCR installation in ERCOT (A) and PJM (B). The vertical black line separates the two weeks. Note that the two weeks are not contiguous, and that the UC model runs them separately.





With a two week period selected using the above algorithm, profits with and without SCR can be calculated. First, profits without SCR for all coal plants are calculated with the UC model by running it for each week in the two week period above. Then, sequentially for each eligible coal plant, SCR installation is simulated and the UC model is run to calculate each plant's profits with SCR. In these runs, only one plant at a time has SCR installation simulated. Plants that have greater profits with than without SCR are assumed to install SCR. A second round of simulations are then made, wherein each coal plant can reverse its prior decision in light of other plants' decisions, such that plants that installed SCR can uninstall it and plants that did not install it can install it. New "baseline" profits for each generator are calculated by running the UC model taking into account first-round SCR installations. Subsequently, for each coal plant in sequence, the prior decision of the plant to install or not to install SCR is reversed and profits are calculated and compared to baseline profits. This process iterates until no coal plants reverse their prior decision, meaning Nash equilibrium is reached.

Given that SCR is dispatchable, plants determine on which days SCR would be run. To make this determination, the operating cost of each power plant – including the cost of emissions – is calculated for each day with and without SCR. If SCR increases a plant's operating cost, SCR is not run on that day. The rationale for this decision is that the operating cost of a single plant does not affect the electricity price when it is not the marginal unit. Thus, higher operating cost would mean lower profits for an infra-marginal unit, which is the normal stack position of a coal plant. For the marginal unit (which is rarely coal), higher operating costs result in no change in profits, as it either stays the marginal unit (and so makes no profits from generation) or is displaced as the marginal unit (in which case it generates nothing).

Profits are calculated as the difference between revenues and costs. Revenue with and without SCR is calculated as electricity generation times the hourly electricity price. The electricity price is obtained from the UC model as the marginal cost of meeting supply with demand. As such, SCR profit calculations include the cost of emissions from a pollution price. Pollution prices need to be included here because the monetary benefit of installing SCR stems from the averted costs of emissions prices. In a cap-and-trade system, for instance, SCR would allow the operator to generate additional revenue by selling permits it would have otherwise "consumed" without SCR. Those permits would be priced, theoretically, at the prevailing emissions price, such as at the time-differentiated price on a high ozone day. Emission permit prices are *not* included when determining system-wide costs, as explained below.

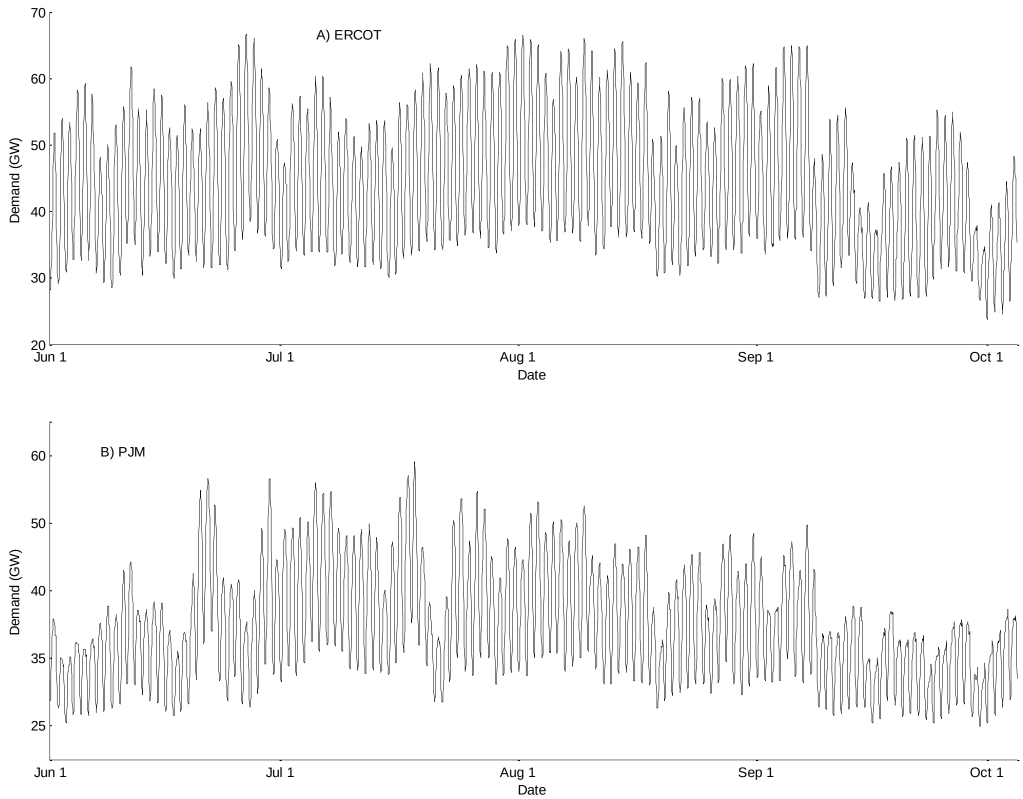
In ERCOT, there is an hourly electricity price (marginal price) for each of the four zones, whereas there is a single system-wide price in PJM. From hourly revenues, which are calculated by multiplying a plant's generation by the electricity price, operational costs are subtracted. Operational costs include fuel, variable O&M, and NOx emissions costs. When no SCR installation is simulated, these operational profits equal total profits. When SCR installation is simulated, however, operational profits are reduced by SCR capital and fixed O&M costs to calculate net profits.

SCR capital and fixed O&M costs on a per-megawatt basis are taken from the EPA's Integrated Planning Model v.4.10 (U.S. Environmental Protection Agency Office of Air and Radiation, 2010). Generator-specific costs are calculated by multiplying EPA's costs by the generator's capacity. Capital costs are annualized over a 15-year period assuming a 6.5% discount rate as in (Sun et al., 2012). Annual costs and fixed O&M were assumed to be paid off entirely over the summer and then scaled down to a two-week period to be comparable to revenues. Annual costs were assumed to be fully paid off over the summer for two reasons. First, our two week period of analysis is representative only of the summer months, so revenues from these two weeks are only representative of summer revenues. Second, current annual NOx permit prices are insufficient to incentivize SCR operation (U.S. Environmental Protection Agency, 2011a). Absent much higher annual NOx permit prices, SCRs would only be used on days when time-differentiated prices trigger, the vast majority of which will be during summer months. Thus, SCR installation must be paid off entirely during the period of time when time-differentiated pricing is relevant, which is the summer.

5.2.3: Second Phase: Determining Summer-Wide Emissions and Costs

After SCR installation decisions are set, system-wide emissions and costs are calculated by running the UC model for each week in the entire summer, from June 1 to October 4. The demand profiles for PJM and ERCOT over this time period are presented in Figure 4 below.

Figure 4: Demand profile of entire summer in 2012 for ERCOT (top) and PJM (bottom).



Total emissions are calculated by summing emissions from power generation and start-ups. Emissions from power generation are calculated as the product of a unit's heat rate (assumed constant in this analysis), power output and the NOx emissions rate. Emissions during start-up are calculated based on the unit's start-up fuel consumption, nameplate capacity and NOx emissions rate. Some units (e.g. open-cycle gas turbines) do not use fuel on start-up in the UC model used here. Additionally, NOx emissions are assumed to be zero from combined-cycle steam turbines, as the steam turbine itself does not generate emissions.

Total system costs equal the sum of dispatching costs and SCR capital costs, if any SCRs are installed. Dispatching costs consist of start-up and operational (power generation) costs. Notably, dispatching costs do not include the cost of emissions that result from time-differentiated prices. The price is only used to signal relative scarcity. If time-differentiated pricing were implemented as a separate allocation of permits or time-varying redemption ratio for use on high-ozone days in a cap-and-trade system, those permits could be allocated at no cost, and electricity prices could be recalculated without pollution prices once dispatching decisions are made. As a result, increased costs to consumers would only be in the form of increased electricity generation costs excluding pollution prices. Alternatively, if assessed as an emissions price, consumers (who would ultimately foot any cost increases from electricity generation) could, in theory, be fully compensated by using the revenue collected from the emissions fees to offset other taxes. Thus, the system cost of a time-differentiated policy results from starting-up and dispatching more expensive generators with lower NOx emissions. The same is true for undifferentiated pricing systems modeled here, which can be equivalently thought of as cap-and-trade systems with free permit allocation or revenue-neutral tax systems.

Two measures of dispatching costs are presented here. One is the "producer cost" – the cost required to run electricity generators dispatched to meet demand. This cost is calculated by multiplying power generation at each generator by the operating cost of that generator. Producer cost is important to consider, as it captures lost economic productivity that may result from a policy. In this case, an emissions price forces substitution towards more expensive but lower-NOx-emitting generators. These more expensive generators may either have more expensive fuel, which indicates more energy is required for their extraction and processing, or be less efficient, meaning they require more fuel input for the same level of output. In either case, more energy is expended for the same output, resulting in economic losses.

The second cost is the "consumer cost" – the cost paid by consumers for electricity. To calculate this cost, total hourly power generation is multiplied by the operating cost of the marginal unit dispatched in that hour, i.e. the marginal cost of electricity. This calculation mirrors pricing of electricity in competitive markets, where all generators are remunerated based on the operating cost of the marginal unit. It therefore captures how a policy may affect consumers through increased costs, a key consideration in assessing policies.

5.3: Analyzed Regulatory Frameworks

Emissions and costs are assessed using the two-phase model for three regulatory schemes: time-differentiated pricing, undifferentiated pricing, and a technology-based standard.

5.3.1: Time-Differentiated Pricing

Time-differentiated pricing is implemented as a cost per ton of emissions on high ozone days. Days are designated as high ozone days if their 24-hour aggregate demand is above a certain threshold. The threshold is determined separately in each power system, and is set such that that the proportion of days that are classified as high ozone days in the two week SCR installation period is equal to the observed proportion of high ozone days in metropolitan areas in the summer of 2012. In ERCOT, four days, or 25% of the two-week SCR installation test period, are classified as high ozone days – roughly representative of the proportion of summer days in which the eight-hour average ozone NAAQS was exceeded in cities like Dallas and Houston (Texas Commission on Environmental Quality, 2013). In PJM, three days of the two-week test period are classified as high ozone, matching the observed proportion in cities like Baltimore, Philadelphia and Washington D.C. (Maryland Department of the Environment, 2012).

Time-differentiated prices tested here range from \$5,000 to \$150,000 per ton. To contextualize the magnitude of these prices, NOx permits under the CAIR, the current cap-and-trade program for NOx, currently trade around \$320 per ton. Also, the average NOx emissions rate at coal-fired generators in the ERCOT dataset used here is roughly 1.3 pounds per megawatt-hour. At a NOx price of \$150,000 per ton, each megawatt-hour of electricity generated from the average coal-fired generator would cost an additional \$100, roughly four times the cost absent an emissions price.

Figure 5 and Figure 6 show the demand profile for the two-week SCR installation test period and the entire summer, respectively, overlaid with which days are classified as high ozone days. Over the entire summer, 22% of days in PJM and ERCOT are classified as high ozone days. High ozone days are not clustered together in the two-week SCR installation test period in either PJM or ERCOT. A sensitivity analysis was performed, though, of whether clustering high ozone days together would alter SCR installation decisions. This analysis was conducted in ERCOT, and the results indicate that SCR installations do not change significantly when all high ozone days are clustered together.

The time-differentiated price on NOx emissions is applied to the entire 24-hour period of high ozone days. If the time-differentiated price is less than the average ozone-season allowance price in 2012 ($20) (U.S. Environmental Protection Agency, 2011c) discounted to 2007 dollars (U.S. Bureau of Labor and Statistics, 2013), then the ozone-season allowance price is used. On non-high ozone days, the same discounted average ozone-season allowance price is used. Each day, the appropriate emissions price is included in the operating cost of each generator by multiplying the emissions price (dollars per ton NOx) by the generator's emissions rate (tons NOx per megawatt-hour generated).

Figure 5: Demand profile (thin black line) for the two week period for determining SCR installations in ERCOT (A) and PJM (B) overlaid with an indicator for which days are high ozone days (black dashed line; high ozone days are those where the line equals 1). In ERCOT, for instance, days 1 and 6 through 8 are high ozone days. The thick black vertical line demarcates the break between the two weeks, as they are run separately in the UC model. High ozone days are determined using a 24-hour aggregate demand threshold.

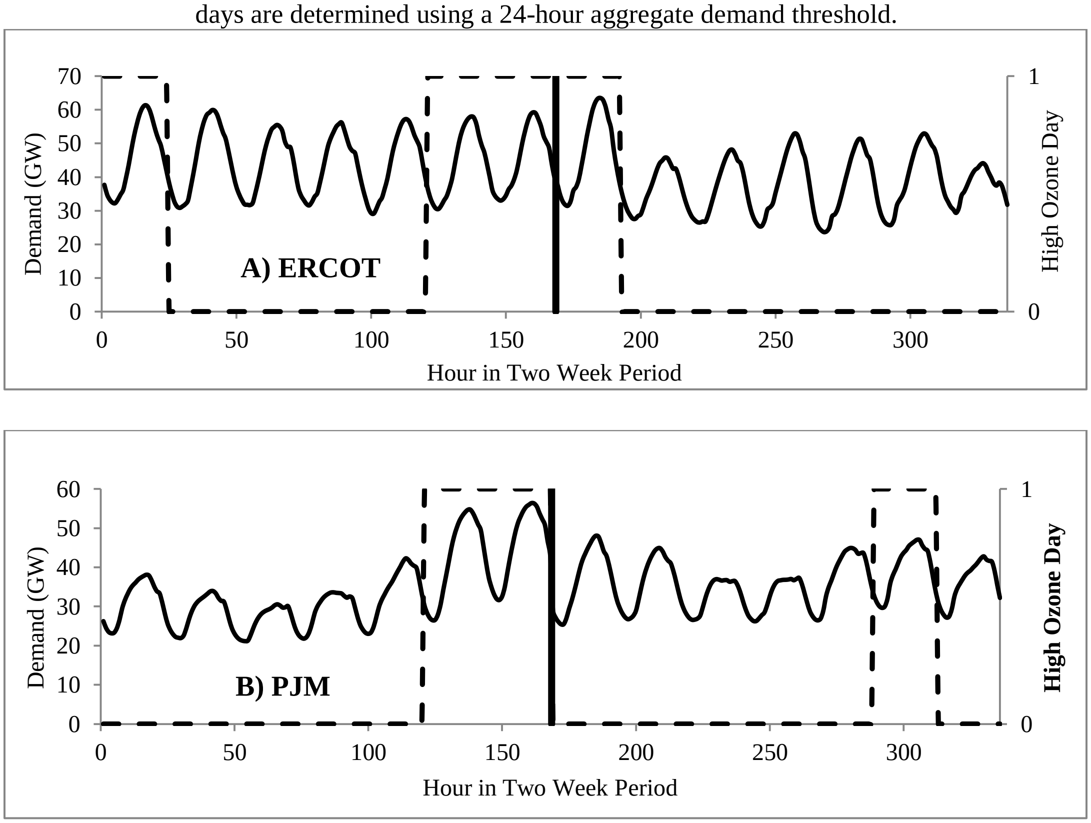
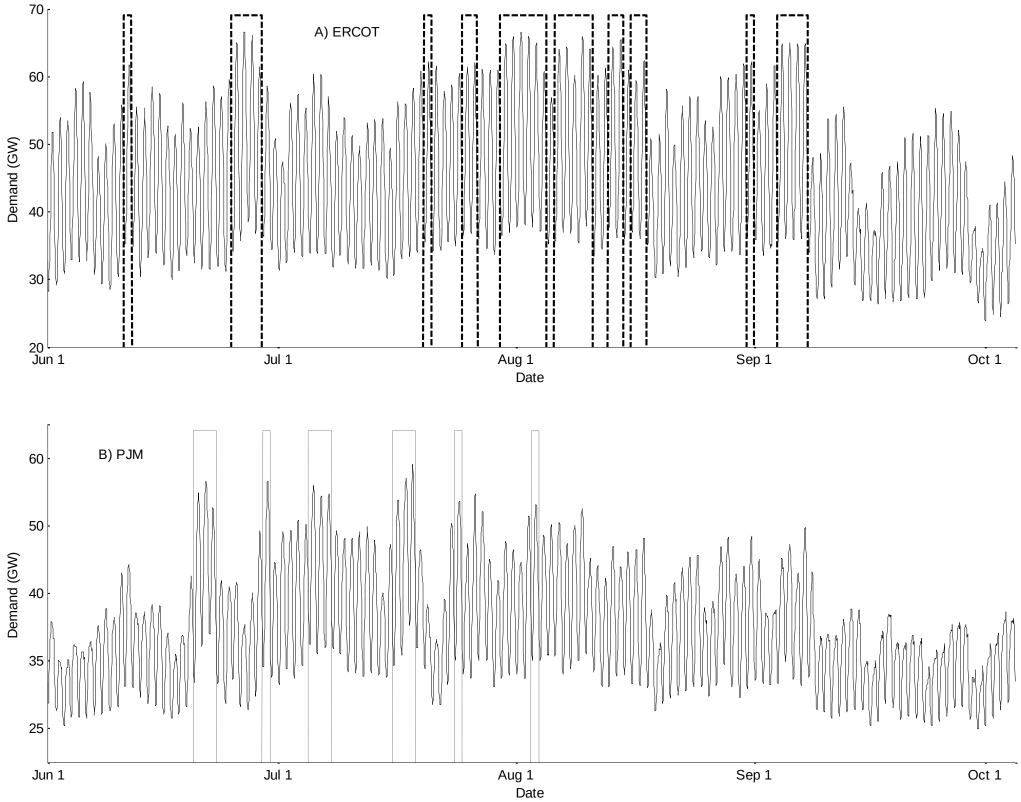


Figure 6: Demand profile over the entire summer in 2012 in ERCOT (A) and PJM (B). Demand that falls under the dashed black line are high ozone days, as determined via a 24-hour aggregate demand threshold.



5.3.2: Undifferentiated Pricing

Undifferentiated prices are also tested here for comparative purposes. In undifferentiated prices modeled here, a flat price is assessed for each ton of NOx emitted over the entire summer. Undifferentiated prices tested here range from $1,000 to $20,000 per ton, as higher prices are determined to not be policy relevant given past undifferentiated prices implicitly set via cap-and-trade programs. (In these programs, emission budgets are typically set such that the marginal cost of abatement is in the hundreds of dollars per ton. No time-differentiated price, on the other hand, has been implemented, so there are no examples to calibrate to.)

In addition to undifferentiated NOx prices, an undifferentiated price scenario is used to represent outcomes that would be achieved under the Cross-State Air Pollution Rule (CSAPR). As discussed in Chapter 3, the CSAPR is the proposed successor to the current cap-and-trade program that covers NOx emissions, the CAIR. Although struck down by the courts, the CSAPR still provides a good estimate of what a future NOx cap-and-trade program would look like. The NOx emissions cap in the CSAPR was set such that the expected marginal cost of reducing annual and ozone-season NOx emissions and SO_2 emissions was $500 per ton (U.S. Environmental Protection Agency, 2011b). As such, an undifferentiated price is modeled here as a $500 per ton permit price on NOx and SO_2 emissions.

5.3.3: Technology-Based Standard

The final type of policy assessed here is a technology-based standard. Under this standard, coal plants must install and operate SCR for the entire summer. This mandate is based on past and present technology-based NOx standards, which require plants abide by a certain emissions rate, which in turn requires the installation of some control technology. The mandate modeled here would force the installation and operation of SCR, either through a direct technology-based standard or an indirect, stringent performance-based standard.

While plants must continually operate SCR when they generate power, plants are not forced to always generate power. Rather, the additional operating cost of SCR is rolled into their operating cost, and then the plants are dispatched by the unit commitment model. Consequently, a plant that is forced to install and operate SCR may not generate power in some hours in the summer when it is not economical.

The stringency levels, or installed capacities, of the SCR scenarios are calibrated to the results of the time-differentiated price scenarios to facilitate comparison. The capacity of SCR to be installed under each technology-based standard is allocated in two different ways. In one case, SCR is installed on those generators that chose to install SCR under the time-differentiated price. This means that SCR is installed on those generators for which SCR installation is most favorable, as it is those generators that choose to install SCR under time-differentiated pricing. Allocating SCR in this manner also provides the best means of comparison between the two policies, and isolates non-control technology effects of time-differentiated pricing. In the second method, SCR is randomly allocated to coal-fired generators until the cumulative capacity of installed SCR is equal to that under the relevant time-differentiated price. This allocation method captures the unpredictably in the implementation of technology-based standards, which are pushed by various forces, e.g. industry and politicians (McCubbins et al., 1987; Stavins, 1998), as discussed at length in Section 3.5. Fifty realizations of randomly-allocated SCRs are modeled in order to capture the range of potential outcomes under a technology-based standard.

Chapter 6: Effect of Time-Differentiated Pricing on Selective Catalytic Reduction Installations

Time-differentiated pricing may affect power plant operations in the long-term by incentivizing the installation of emission control technologies. The two-phase model developed for this research captures this potential response, specifically of coal-fired generators installing SCR. SCR installations are only observed at very high time-differentiated prices. At these prices, a moderate number of SCRs is installed in ERCOT (5 SCRs), whereas many are installed in PJM (23 SCRs). SCR installation decisions are shown to be driven mainly by three factors: each generator's NOx emissions rate and generation profile *without* SCR as well as the minimum achievable NOx emissions rate that can be achieved with SCR. Finally, two sensitivity analyses are run on SCR installation decisions. The first shows that clustering high ozone days does not significantly change SCR installation decisions. The second shows that using a simplified power system model underestimates SCR installations.

6.1: SCR Installations in Response to Time-Differentiated Pricing

6.1.1: ERCOT

In ERCOT, very high time-differentiated prices are required before coal-fired generators begin to install SCR. No SCR installations are observed at time-differentiated prices below \$125,000 per ton of NOx in ERCOT. Three coal generators (of a possible 19) install SCR at a NOx price of \$125,000 per ton, and five install it at \$150,000 per ton. The cumulative capacity of coal-fired generators that decide to install SCR is 2.5 GW at \$125,000 and 4.5 GW at \$150,000.

At Nash equilibrium, every generator adopts a mixed strategy in regards to SCR installation. When no other generators install SCR, SCR installation is profitable for certain generators, so they install SCR. With SCR, those generators can produce power on high ozone days, but this depresses prices on those days. Lower prices, in turn, make SCR installation less or not profitable. In this case, SCR installation becomes not profitable at every generator when other generators install SCR as determined here. Consequently, when allowed to reconsider SCR installation decisions in light of other generators' decisions, this price signal is conveyed, and all generators decide not to install SCR. But each generator is then allowed to reassess its new decision in light of other generators' new decisions, at which point each generator that chose to install SCR before again chooses to install SCR. This mixed strategy is adopted by all generators that choose to install SCR in ERCOT; no generators adopt a pure strategy of SCR installation, i.e. install SCR regardless of other generators' decisions.

One contributing factor to the adoption of mixed strategies by every generator is likely the lumpiness of SCR investment. Generators are only permitted to install and operate SCR for the plant's entire power capacity. In other words, a 500 MW plant must install and operate SCR on the entirety of its 500 MW potential output, rather than a fraction of it. The lumpy nature of SCR installations has two effects. First, each SCR has a larger impact on system-wide prices than it might otherwise. Second, generators are not permitted to install smaller increments of SCR that may be profitable even when SCR on the entire plant is not.

6.1.2: PJM

A similar trend of an increasing number and capacity of installed SCR with increasing time-differentiated prices occurs in PJM as in ERCOT (Table 3). However, PJM and ERCOT differ greatly in the number of SCRs installed and the NOx price at which SCRs are installed. In PJM, generators begin to install SCR at a price of $75,000 per ton, versus $125,000 per ton in ERCOT. Additionally, many more plants install SCR in PJM than ERCOT. At $75,000 per ton, seven plants with a combined capacity of 4 GW install SCR, about the same as the installed SCR capacity at $150,000 per ton in ERCOT. At $125,000 and $150,000 per ton in PJM, 23 plants install SCR with a cumulative capacity of 7 GW. While nearly five times as many SCRs are installed at a $150,000 price in PJM as in ERCOT, installed capacity in PJM is less than double that in ERCOT because the coal-fired generators are smaller in PJM than ERCOT.

Table 3: Number of coal-fired generators that install SCR and cumulative capacity of installed SCR at time-differentiated prices in PJM. No SCRs are installed at prices at or below $50,000 per ton.

Time-Differentiated NOx Price (thousand $/ton)	Number of Coal-Fired Generators that Install SCR	Cumulative Capacity of Installed SCRs (GW)
50	0	0
75	7	4
100	11	5
125	23	7
150	23	7

Unlike in ERCOT where some plants adopt a mixed strategy, in PJM no plants change their SCR installation decisions in light of other plants' decisions, except for one additional plant that installs SCR at $75,000 per ton. Quantities of SCR installations in PJM are similar to those documented by (Bharvirkar et al., 2004), who found post-combustion control technologies, including SCR, would be installed at all coal plants in Maryland in response to a time-differentiated pricing scheme. Here, 60% of coal plants install SCR in PJM Classic at a time-differentiated price of $150,000 per ton, a significant share.

6.2: SCR Installations in Response to Undifferentiated Policies and the CSAPR

No SCRs are installed in any of the undifferentiated policies tested here, which range from $500 to $20,000 per ton. The result at $500 per ton is consistent with analysis by the EPA, which projected that no SCRs would be installed as a result of the CSAPR (U.S. Environmental Protection Agency, 2011b, Table VII.C.2-1), under which NOx permit prices were expected to be $500 per ton. Section 6.3.3 below explains why no SCR installations are observed under any undifferentiated price.

6.3: Factors Affecting SCR Installations

While myriad factors may affect a generator's decision to install SCR, three factors seem to dominate installation decisions here: the generator's NOx emissions rate, its power generation profile *without* SCR, and the minimum NOx emissions rate achievable with SCR installed.

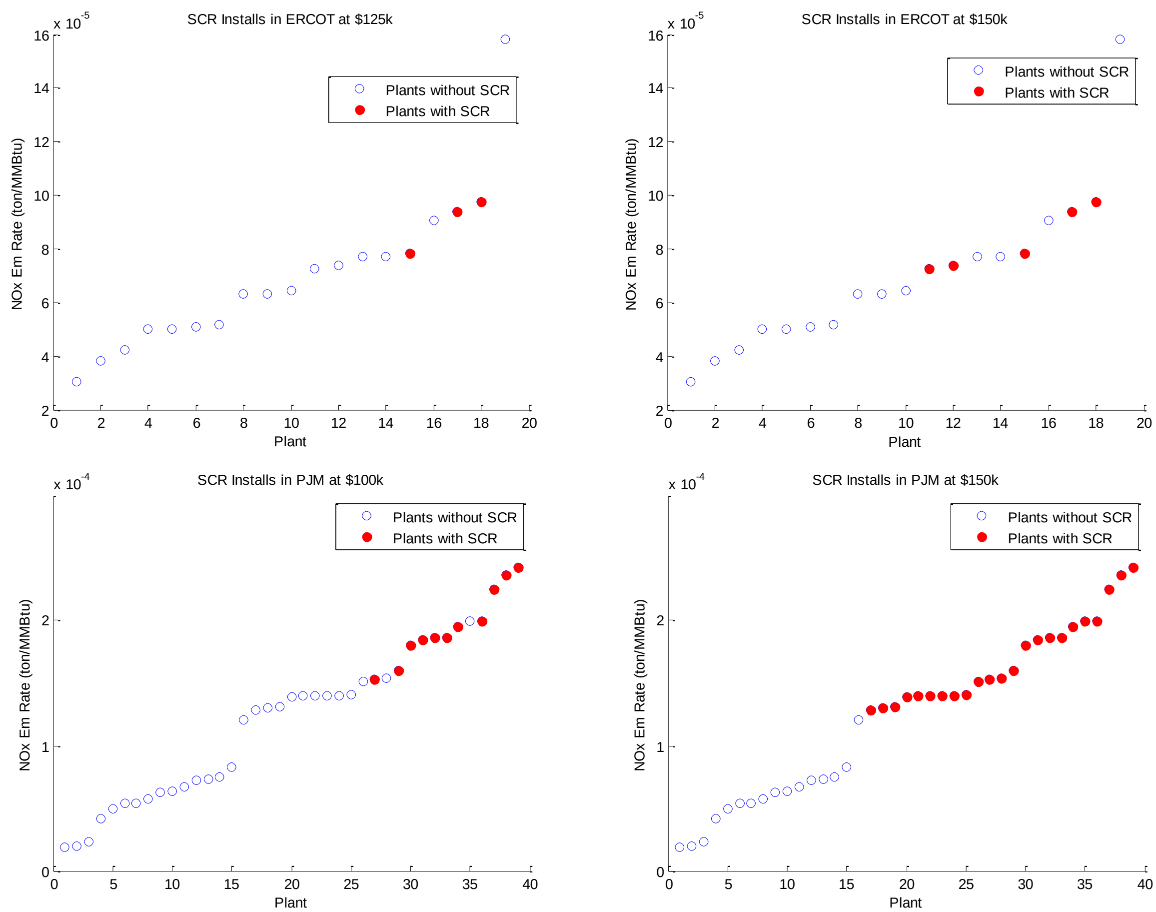
6.3.1: NOx Emissions Rate

A generator's initial NOx emissions rate strongly impacts its decision to install SCR because SCR cannot reduce a generator's NOx emissions rate below a certain threshold. The profitability of SCR is driven by how much it reduces NOx emissions, which in turn reduces operating costs and increases operational profits. Thus, the lower the reduction in NOx emissions from SCR operation, the lower its potential profitability. Here, no coal-fired generator in ERCOT or PJM tested for SCR installation achieves the full potential 90% emissions rate reduction from

SCR installation; all of the generators that install SCR hit the emissions rate floor achievable with SCR. Thus, those generators with higher initial NOx emissions rate get greater emission reductions – and therefore greater potential profits – from SCR installation.

The strong relationship between NOx emissions rate and SCR installations in both PJM and ERCOT is demonstrated in Figure 7. Plants that choose to install SCR (filled circles) tend to have higher NOx emissions rates than plants that choose not to install SCR (open circles). This relationship holds true at low time-differentiated prices when SCR installations are first observed in each power system, as well as at higher prices when more SCRs are installed. The relationship is especially prominent in PJM, where at a $100,000 time-differentiated price, 11 of the 12 plants with the highest NOx emissions rates install SCR, and at a $150,000 time-differentiated price, the 23 plants with the highest NOx emissions rates all install SCR. The reason that this relationship is more prominent in PJM than in ERCOT is related to the next factor that drives SCR installations, operations without SCR.

Figure 7: NOx emissions rates of coal-fired generators that install (filled circles) and do not install (open circles) SCR in ERCOT (top) and PJM (bottom) under time-differentiated prices. Results from two time-differentiated prices for each system are shown: in ERCOT, $125,000 (left) and $150,000 (right), and in PJM, $100,000 (left) and $150,000 (right).



6.3.2: Operations without SCR

A generator's decision to install SCR is also strongly influenced by its operations *without* SCR. In this research, those operations are determined by imposing a certain policy, such as time-differentiated pricing, and running the unit commitment model before allowing generators to install SCR. Two possible operational "modes" exist for generators prior to SCR installations: it is either dispatched or not dispatched on days with an emissions price (e.g., on high ozone days in the case of time-differentiated pricing). Installing SCR has different benefits in each operational mode.

For generators that are not dispatched on days with an emissions price, installing SCR may lead to the generator being dispatched on those days by reducing its operational cost. While many factors go into dispatching decisions in a unit commitment model, operational costs are a significant factor – generally, cheaper plants are dispatched first. The fact that a unit is not dispatched in a unit commitment model suggests that it is too expensive to run. Under a NOx emissions price, installing SCR reduces a generator's NOx emissions, which in turn reduces the plants overall operating cost (despite the additional cost required to operate SCR). Consequently, the generator becomes more favorable for dispatching during hours with an emissions price. If the generator is dispatched, SCR installation has essentially extended the generator's operational hours and allowed it to generate profits on hours it otherwise would not be able to. For such generators, then, SCR will only be installed if it significantly reduces operational costs, such that the plant is dispatched when it otherwise would not be. Because reductions in operational costs are driven by reductions in NOx emissions, the key driver of SCR installation for these generators is their initial NOx emissions rate.

In the case of generators that are dispatched on days with an active emissions price even without SCR, SCR serves primarily as a cost reduction tool. Installing SCR for these generators will not significantly increase the number of operational hours. Rather, installing SCR will reduce the operational costs of a generator when it is already generating power, which increases profits. For generators in this operational mode, the NOx emissions rate is only important insofar as it is high enough to guarantee recovery of SCR fixed costs through operational cost reductions.

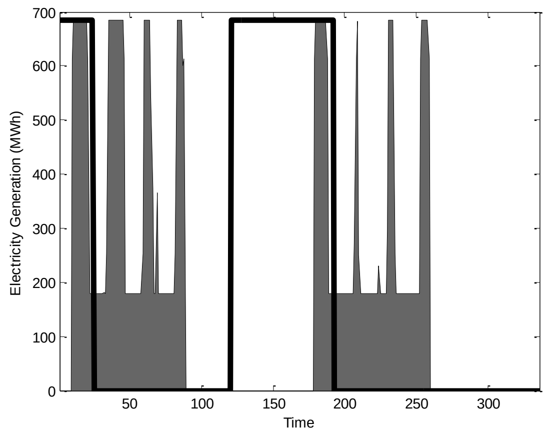
6.3.3: Minimum Achievable Emissions Rate with SCR

NOx emissions rate at coal-fired generators cannot be reduced beyond a minimum threshold even with SCR installed. The minimum threshold is based on SCR parameters used by the EPA (U.S. Environmental Protection Agency Office of Air and Radiation, 2010). Because of this threshold, coal-fired generators cannot reduce their NOx emissions rates below those of many gas-fired generators even with SCR. In ERCOT, the lowest NOx emissions rate achievable at a coal-fired generator with SCR installed is roughly 0.5 lb/MWh; roughly 34 GW of gas-fired capacity exists in ERCOT with a NOx emissions rate lower than 0.5 lb/MWh. On top of that gas-fired capacity is another 5 GW of nuclear, which has no NOx emissions, and other zero-NOx electricity sources like wind and solar. In PJM, there's less gas-fired capacity (12 GW) with a lower NOx emissions rate than the lowest achievable rate at a coal-fired generator with SCR (0.33 lb/MWh), but more nuclear capacity (15 GW).

Natural gas and other plants with lower NOx emissions rates may be cheaper to run when a NOx emissions price is active than a coal-fired generator with SCR installed. At a $20,000 per ton undifferentiated price on NOx emissions in ERCOT, for instance, over 30 GW of gas-fired capacity is available that's cheaper to operate than any coal-fired generator with SCR that's tested for SCR installation. Thus, even by installing SCR, a coal-fired generator may not necessarily generate power in every hour when a price on NOx emissions exists because generators with lower NOx emissions rates may be available and cheaper.

This factor is particularly important in explaining the lack of SCR installations under undifferentiated pricing. In many low-demand hours in the summer, demand can be met entirely by generators with lower NOx emissions rates than a coal-fired generator with SCR can achieve. As such, even by installing SCR, a coal-fired generator may only be economic to operate under an undifferentiated price for a small number of hours over the summer when demand is high. An example of this generation profile is provided in Figure 8 for a coal-fired generator with SCR installation simulated under a $20,000 per ton undifferentiated price in ERCOT. The coal-fired generator does not generate over many hours during the two week period, and even does not generate in each hour of every high ozone day.

Figure 8: Power generation of a coal-fired generator with SCR installed over two-week SCR installation test period under a $20,000 per ton undifferentiated price in ERCOT. The thick black line denotes hours of high ozone demand.



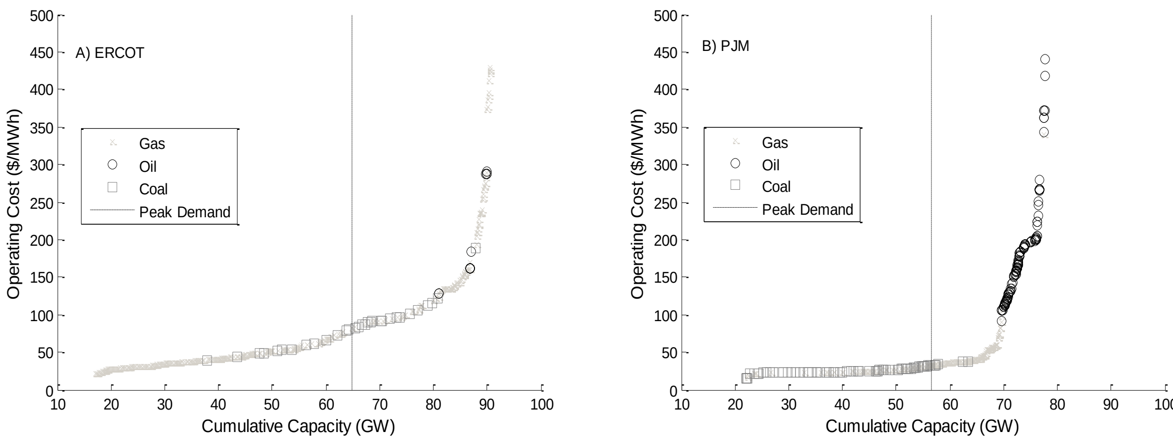
Without SCR, the same coal-fired generator does not produce any electricity in this two week period. So, although the generator does not produce at maximum output over the entire two-week period with SCR, it does produce more power than it would otherwise. However, profits with SCR as calculated here include annualized capital costs. SCR is only installed if operational profits with SCR minus capital costs are greater than profits without SCR. In this case, while operational profits are greater with SCR than without (as the latter are zero), operational profits with SCR are less than the annualized capital cost of SCR. As such, the plant would *lose* money by installing SCR, so it does not. A similar situation plays out for all generators tested for SCR installation in ERCOT and PJM; operational profits with SCR are greater than without SCR for every generator (note that some generators do have positive profits even without SCR), but when annualized capital costs of SCR are accounted for, installing SCR would result in losses at every generator.

6.4: Comparison of SCR Installations in PJM and ERCOT

Higher SCR installation rates in PJM are attributable to differences in the operational modes of coal-fired plants between the power systems. For one, some generators install SCR in order to generate on high ozone days in PJM, but not ERCOT. Additionally, while coal-fired generators install SCR because they are dispatched on high ozone days even without SCR in both systems, far more coal plants are dispatched in this manner in PJM than in ERCOT. In both cases, SCR installation is profitable for more plants in PJM than in ERCOT.

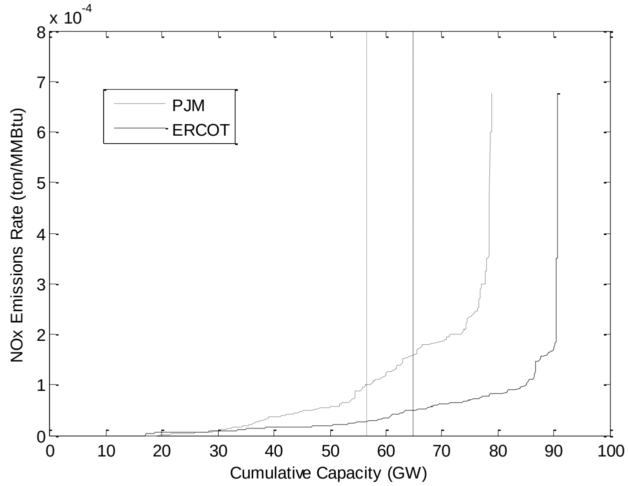
In regards to the latter case, more coal plants are dispatched on high ozone days even without SCR in PJM for two reasons. First, coal-fired generation is needed in PJM, but not necessarily ERCOT, to meet peak demand during the two-week SCR installation period. Whereas non-coal capacity in ERCOT is larger than peak demand, non-coal capacity in PJM is roughly 2 GW less than peak demand and is less than demand for 13 hours. (Peak demand for both systems is on a high ozone day.) Second, there is much more low-cost and low-NOx-emitting gas-fired generation in ERCOT than in PJM (Figure 9). These gas-fired generators are cheap to operate even on high ozone days, since a NOx emissions price largely does not change their operational cost. As such, these gas-fired generators can meet a higher share of demand on high ozone days in ERCOT, reducing the need to run coal-fired generators on those days.

Figure 9: Operating cost curves of electricity generators by fuel type in ERCOT (A) and PJM (B). Dashed lines show the peak demand in each system in the two-week period over which SCR installation is tested.



The second reason why more coal plants install SCR in PJM – specifically, that coal plants install SCR in order to operate on high ozone days in PJM but not ERCOT – is attributable to higher NOx emissions rates in PJM. The NOx emissions rate curves for PJM and ERCOT are shown in Figure 10. The emissions curve for PJM is much steeper than for ERCOT and also increases closer to PJM peak demand (dashed line), indicating that marginal electricity prices will increase quicker in PJM under emissions pricing. This is especially true at high pollution prices, when the cost of emissions dominate all other operating costs. Higher electricity prices, in turn, mean infra-marginal generation will generate more profits, creating a larger potential for coal plants to move infra-marginally and generate sufficient profits to cover fixed SCR costs.

Figure 10: Emissions rate curves for PJM (gray line) and ERCOT (black line). The curve traces NOx emissions rate of generators in each system against their cumulative power capacity. Vertical dashed lines indicate peak demand over the two-week SCR installation test period in PJM (gray) and ERCOT (black).



6.5: Sensitivity Analyses for SCR Installations

Two sensitivity analyses were run on SCR installation decisions in ERCOT to test how high ozone day clustering and the type of power system model used affect SCR installations. The former analysis suggests that SCR installations as determined here are robust to whether high ozone days are clustered or distributed throughout the two-week installation test period. The latter analysis demonstrates that a simplified power system model that neglects real-world operational constraints on power plants underestimates SCR installations.

6.5.1: Clustering of High Ozone Days

The sensitivity of SCR installations in ERCOT to the grouping of high ozone days (on which the time-differentiated price is triggered) in the two-week SCR installation test period was examined. Rather than having three distinct “ozone episodes” in the two-week period, all four high ozone days were grouped into a single “ozone episode” at the end of the first week. Two of the four days at the end of the first week already qualified as high ozone days based on their daily demand, and the preceding two days were also high demand days.

This sensitivity analysis was run for two reasons. First, it accounts for potential impacts of the inflexibility of coal plants. Coal plants are constrained in how quickly they can ramp, i.e. change their power output, as well as in their minimum power output. They also have high start-up costs. Thus, coal plants are best suited to steady power generation. Clustering high ozone days makes operations more amenable to steady levels of power generation, both before and during high ozone days.

Second, the sensitivity accounts for the fact that coal plants would likely be operating at full output when a high ozone day is triggered. In this sensitivity analysis, all high ozone days are in the first week, and are preceded by three normal days. These three days simulate a normal stretch of time preceding the high ozone day. Conversely, when basing high ozone days based on demand thresholds, each week begins with a high ozone day. Thus, there is no stretch of normal operations in which coal plants may be run prior to time-differentiated prices triggering. This may distort SCR installation decisions, such as through increased start-up costs.

SCR installations in the aggregate do not significantly change with clustering of high ozone days. Pre-Nash installation decisions are identical with and without high ozone clustering at $125,000 and $150,000 prices. Once Nash equilibrium is reached at a $125,000 price, one fewer plant installs SCR when high ozone days are grouped. At $150,000 per ton, the same number of SCRs – five – is installed in both cases, and four of those are at the same plants.

These results suggest SCR installation decisions presented above are robust across different temporal patterns of high ozone days across the two-week testing period. They also suggest that in the real-world, SCR installation decisions may be fairly insensitive to uncertainty in the temporal distribution of high ozone days across summers.

6.5.2: Sensitivity to Power System Model Used in Determining SCR Installations

SCR installation decisions were also tested in ERCOT and PJM under a simpler power system model that neglects real-world operating constraints on power plants. This simpler model, termed an economic dispatch model, meets demand at least cost while ignoring generator ramping and minimum load constraints and start-up costs. It also does not account for spinning reserves. Thus, dispatching of power plants obtained using an economic dispatch is often infeasible in the real world. Notably, prior research (Bharvirkar et al., 2004; Sun et al., 2012) on time-differentiated pricing of NOx emissions on high ozone days has used economic dispatch models.

With an economic dispatch model, fewer SCR installations are projected, emphasizing the need for a more operationally-detailed power system model. At a $150,000 price in ERCOT, only two plants install SCR when an economic dispatch model is used, compared to five using a unit commitment model. Similarly, at a $125,000 price, two plants installed SCR when an economic dispatch is used, one less than with a unit commitment model. In PJM, a similar trend emerged: at a $125,000 price, 23 units installed SCR with unit commitment, whereas only 12 did with economic dispatch.

Lower installation rates are attributable to the lack of constraints in an economic dispatch model. In a unit commitment model, starting-up coal-fired generators incurs a cost, so keeping them operating through high ozone days, while expensive, may ultimately be cheaper. Additionally, coal plants may need to be run on high ozone days to meet peak demand; in a unit commitment model, as in the real world, coal plants could not be start-up and shut-down just on those hours to serve peak demand. In these instances, SCR installation may make sense because it reduces the costs of operating a coal plant during the high ozone day. Conversely, in an economic dispatch model, starting-up coal plants does not incur a cost, and they can be switched on and off just in peak demand hours in order to meet demand. Thus, coal plants will (unrealistically) be run less by an economic dispatch model on high ozone days when they are heavily penalized for emissions, reducing SCR installation incentives.

6.6: Summary and Discussion

Time-differentiated pricing of NOx emissions on high ozone days can lead to SCR installations at coal-fired generators, but only at very high prices. In ERCOT and PJM, the highest price tested here – $150,000 per ton – led to 4.5 and 7 GW of SCR installations, respectively, a sizable portion of coal-fired capacity in each system. At moderate and low time-differentiated prices (e.g., prices below $75,000) no SCR installations were observed. Thus, at these prices, time-differentiated pricing will reduce NOx emissions only through short-term effects on power plant operations, mainly redispatching.

SCR installations are driven by numerous factors, but the dominant factors observed here are a generator's initial NOx emissions rates, its operational mode *without* SCR, and the minimum achievable NOx emissions rate with SCR. The former two factors explain differences in observed installations in PJM and ERCOT; in general, power systems with greater coal-fired generation, like PJM, will have more SCR installations in response to a time-differentiated price than a system that is less reliant on coal-fired generation, like ERCOT. The latter factor explains why no SCR installations are observed under undifferentiated prices tested here; a large capacity of gas-fired generators, as well as nuclear, wind and solar generators, have lower NOx emissions rates than what coal-fired generators can achieve even with SCR. As such, these lower-NOx-emitting generators are often dispatched first when a NOx price is active, meaning a coal-fired generator would not be dispatched even with SCR for low-demand hours in the summer.

Chapter 7: System-Wide Emissions and Costs under Time-Differentiated Pricing

This chapter presents system-wide costs and emissions under various levels of time-differentiated pricing in PJM and ERCOT. Costs and emissions reductions are presented for high ozone days, which is the primary focus of the time-differentiated pricing regime analyzed here. They are also presented for the entire summer for two reasons: reducing summer-wide emissions is also a priority of emissions policies, and emissions may change on days other than high ozone days due to generator operational features like start-up emissions, ramping and minimum load constraints. Factors that affect emissions and costs under time-differentiated pricing are also explored, including why outcomes differ between PJM and ERCOT. Unless explicitly indicated otherwise, costs presented below are producer costs, calculated as the product of each generator's electricity generation and operating cost plus start-up and fixed costs of SCR installations (if any).

7.1: Time-Differentiated Pricing in ERCOT

7.1.1: Emissions and Costs on High Ozone Days

Time-differentiated pricing can significantly reduce NOx emissions on high ozone days in ERCOT, but average reduction costs increase with time-differentiated prices (Figure 11). At the lowest time-differentiated prices explored here, $5,000 and $10,000 per ton, NOx emissions fall by a substantial 15% and 22%, respectively. NOx emissions decline by half at the highest price of $150,000 per ton. Total costs increase with NOx price, as do marginal costs. Costs increase by 0.4% at $5,000 per ton, but increase by 13% at $150,000 per ton. Consequently, the average cost of emissions reductions – defined here as the cost per ton of NOx emissions reduced – increases with increasing time-differentiated price (Table 4). For instance, the average ton of NOx reduced at $150,000 per ton is an order of magnitude more costly than the average ton at $5,000 per ton.

Figure 11: Total cost versus NOx emissions on high ozone days in ERCOT under various time-differentiated prices. Labels next to each point indicate the time-differentiated price assessed per ton of NOx emissions.

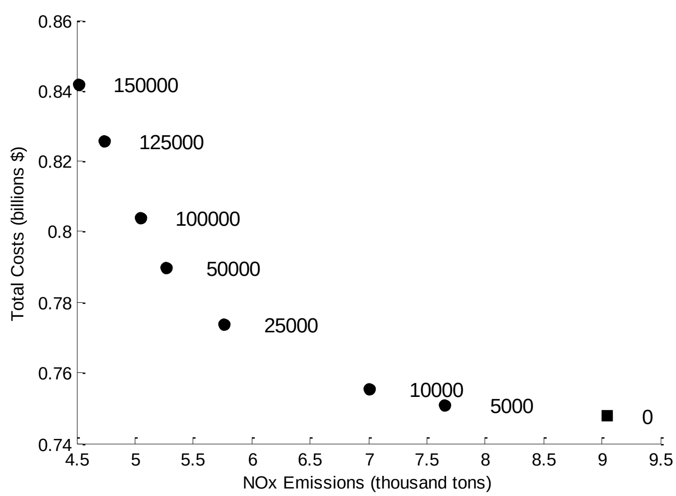
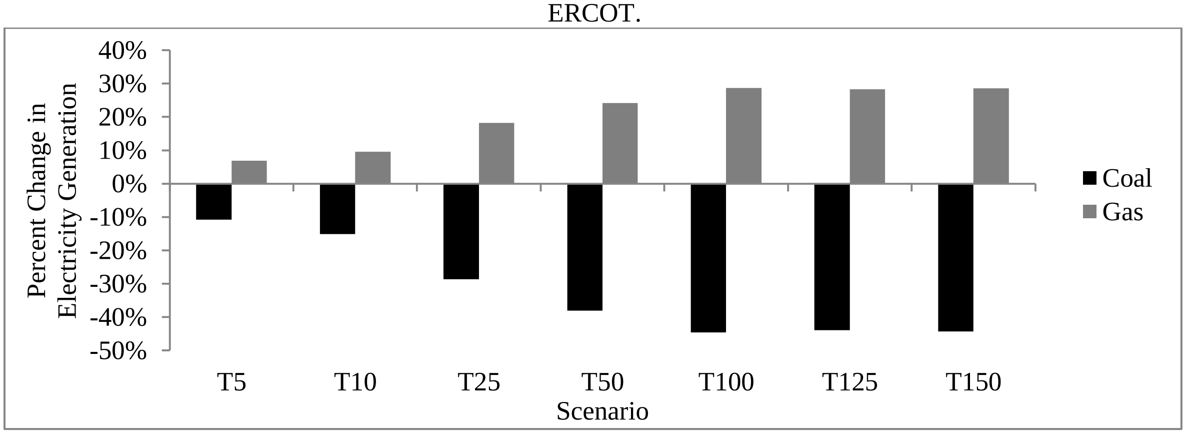


Table 4: Cost and emissions on high ozone days in ERCOT under various time-differentiated prices.

Time-Differentiated NOx Price (thousand $/ton)	NOx Emissions (thousand tons)	Total Cost (million $)	NOx Emissions Reduced (thousand tons)	Total Cost Increase (million $)	Average Cost per Ton NOx Reduced (thousand $/ton)
0	9.04	747.92	0.00	0.00	N/A
5	7.65	750.88	1.39	2.96	2.13
10	7.01	755.54	2.03	7.62	3.74
25	5.76	773.86	3.28	25.94	7.90
50	5.26	790.03	3.78	42.11	11.15
100	5.04	804.07	4.00	56.14	14.05
125	4.74	825.98	4.30	78.06	18.14
150	4.52	841.88	4.52	93.96	20.79

Emission reductions are driven primarily by substituting gas- for coal-fired generation. Nuclear electricity generation does not change in any price tested here and oil accounts for zero or a negligible fraction of total generation. Figure 12 shows that as prices increase, coal-fired generation decreases and is replaced by gas-fired generation. Significant redispatching occurs even at low prices; at $10,000 per ton, for instance, gas-fired generation supplants roughly 20% of coal-fired generation. At high prices ($100,000 per ton and above), gas-fired generation displaces roughly half of coal-fired generation. Consequently, whereas under the baseline scenario coal and gas account for 35% and 55% of electricity generation, respectively, at a $150,000 per ton time-differentiated price gas accounts for 70% of electricity generation while coal only accounts for 19%. Redispatching occurs because gas-fired facilities have much lower NOx emissions rates than coal-fired generators. Consequently, a price on NOx emissions imposes a much greater cost on coal- than on gas-fired generation, leading to gas-fired generators being dispatched instead of coal-fired generators.

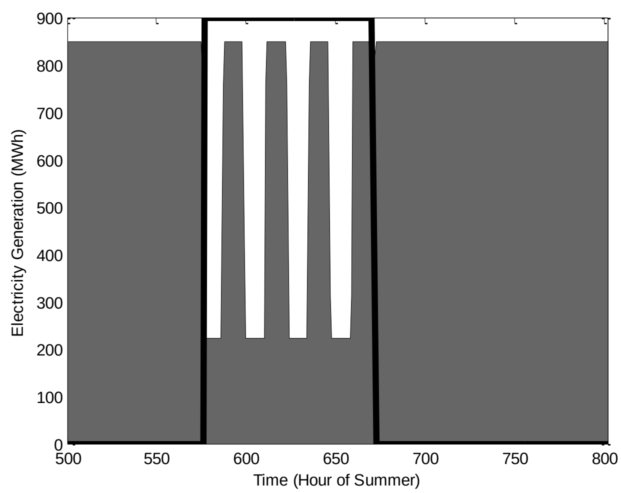
Figure 12: Percent change from baseline (no time-differentiated price) of electricity generated by coal-fired (black) and gas-fired (gray) generators on high ozone days across time-differentiated prices (labeled in thousand $/ton) in ERCOT.



Electricity generation from coal-fired generators flattens out above $100,000 per ton as generators begin to install SCR. By installing SCR, a coal-fired generator mitigates most of the operational cost increase incurred under NOx prices. Those plants that install SCR are therefore able to compete with gas-fired generation and produce power on high ozone days. Notably, although coal-fired generation remains flat, NOx emissions continue to decline at prices of $125,000 and $150,000 per ton because an increasing share of coal-fired generation occurs with SCR operating. This means that while the bulk of emission reductions at these high prices stem from redispatching, the *additional* emission reductions at prices of $125,000 and $150,000 per ton arise from the installation and operation of SCR.

Total generation from coal-fired generators does not increase when SCRs are installed for two reasons. First, coal-fired generators do not operate at maximum output for every hour on high ozone days even with SCR installed. Figure 13 is a representative generation profile of a coal-fired generator with SCR installed over a subset of summer hours at a $150,000 per ton time-differentiated price in ERCOT. The generator reduces its power output on off-peak hours on high ozone days because, as explain in Section 6.3.3, a significant capacity of gas-fired and other generators are available in ERCOT that have lower NOx emissions than the coal plant even with SCR. These lower-NOx-emitting generators are cheaper to operate on hours when an emissions price is active, so on off-peak hours when these generators can meet demand, the coal-fired generator is not dispatched at its maximum capacity. For the same reason, coal-fired generators with SCR installed also do not operate in every hour of every high ozone day over the summer. Rather, on average, coal-fired generators with SCR operate roughly 90% of the hours on high ozone days.

Figure 13: Electricity generation at a coal-fired generator that chooses to install SCR at a $150,000 time-differentiated price in ERCOT. The thick black line indicates hours on high ozone days when the time-differentiated price is active.



The second reason why total coal-fired generation does not increase with more SCRs installed is that SCR is only installed in ERCOT by generators that are already producing power on high ozone days, as discussed in Section 6.4,. Thus, SCR installations in ERCOT are not adopted by coal-fired generators to begin producing power on high ozone days. These two facts combined indicate that SCR installations would not significantly increase coal-fired generation on high ozone days in ERCOT.

Total cost on high ozone days is broken down by component in Table 5 below. Energy accounts for the vast majority, 95% or more, of total costs at each price. Start-up costs, which include fixed and fuel costs, constitute a small and largely-static share of total costs. Fixed costs from SCR installations similarly account for just a tiny fraction of total costs when SCR is installed. Costs increase with increasing time-differentiated prices because of increasing energy costs. Increasing energy costs are driven by redispatching of previously more expensive (gas-fired) generation for previously less expensive (coal-fired) generation.

Table 5: Total and disaggregated cost in ERCOT on high ozone days across time-differentiated prices.

Time-Differentiated NOx Price (thousand $/ton)	Total Cost (million $)	Energy Cost (million $)	SCR Fixed Cost (million $)	Start-up Cost (million $)
0	747.92	728.05	0.00	19.88
5	750.88	730.74	0.00	20.14
10	755.54	735.18	0.00	20.35
25	773.86	752.83	0.00	21.04
50	790.03	768.54	0.00	21.49
100	804.07	781.66	0.00	22.40
125	825.98	793.38	9.85	22.76
150	841.88	801.98	16.36	23.55

7.1.2: Emissions and Costs for Whole Summer

Emission reductions on high ozone days and aggregate reductions over the whole summer exhibit different patterns. NOx emission reductions essentially only occur on high ozone days under time-differentiated pricing. At lower prices, emissions reductions over the entire summer are actually slightly less than those just on high ozone days. Emissions during start-ups of coal-fired generators after high ozone days could account for this disparity; these emissions would offset some of the reductions obtained on high ozone days by shutting off coal-fired generators, leading to slightly greater summer-wide emissions.

Overall, the proportion of summer-wide NOx emissions reduced under time-differentiated pricing is much less than the proportion on high ozone days. Summer-wide emission reductions range from 5% to 13% for the prices explored here (Figure 14). The average cost per ton of these reductions is also greater than on high ozone days (Table 6). As on high ozone days, the average cost per ton of NOx emissions reduced increases with price; more than an order of magnitude separates the average cost of emission reductions at $5,000 and $150,000 per ton. These costs are driven overwhelmingly by energy costs; start-up and fixed costs constitute a negligible fraction of total costs.

Figure 14: Total cost versus NOx emissions over the entire summer in ERCOT under time-differentiated prices (circles, labeled in $/ton).

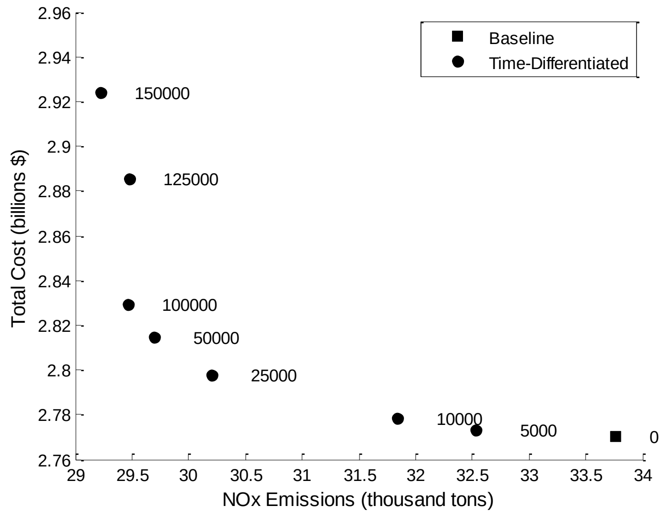
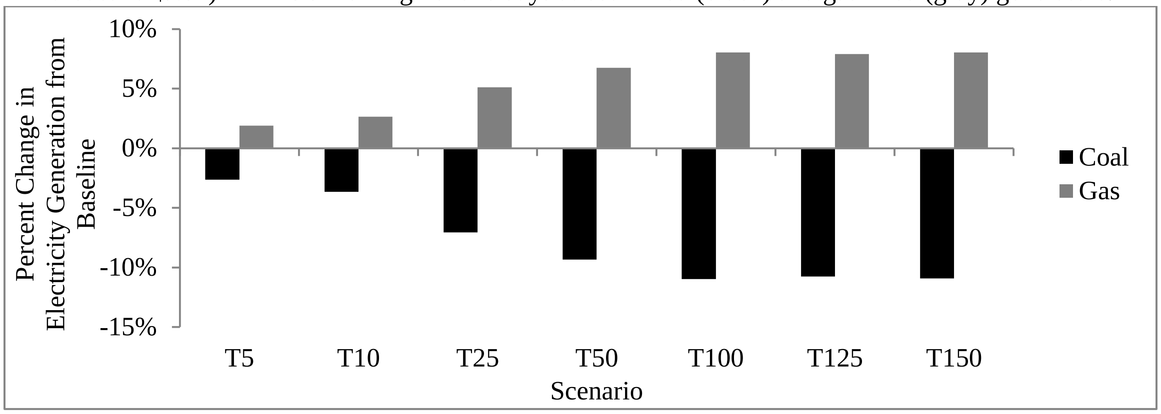


Table 6: Costs and emissions over the entire summer in ERCOT under time-differentiated prices.

Time-Differentiated NOx Price (thousand $/ton)	NOx Emissions (thousand tons)	Total Cost (million $)	NOx Emissions Reduced (thousand tons)	Total Cost Increase (million $)	Average Cost per Ton NOx Reduced (thousand $/ton)
0	33.77	2770.28	0.00	0.00	N/A
5	32.53	2773.37	1.23	3.09	2.51
10	31.84	2778.37	1.93	8.09	4.20
25	30.21	2797.54	3.56	27.27	7.66
50	29.69	2814.49	4.07	44.21	10.85
100	29.47	2829.26	4.29	58.98	13.73
125	29.47	2885.36	4.29	115.08	26.81
150	29.23	2924.23	4.54	153.95	33.94

Substitution of gas- for coal-fired generation is still evident over the entire summer, as shown in Figure 15 below, but to a lesser extent than just on high ozone days. At $150,000 per ton, for instance, coal generation over the entire summer decreases by around 11%, compared to a 45% decrease on just high ozone days. This indicates that coal-fired generators continue to provide baseload generation on non-high ozone days when the time-differentiated price is not active.

Figure 15: Percent change in electricity generation from baseline under time-differentiated prices (labeled in thousand $/ton) in ERCOT on high ozone days at coal-fired (black) and gas-fired (gray) generators.



Non-served energy results indicate that generators in ERCOT can meet demand without hitting the market price cap under even the highest time-differentiated prices tested here. While non-served energy is not included in the unit commitment (UC) model when determining SCR installations, it is included in the UC model when determining dispatching over the entire summer to calculate emissions and costs. NSE is priced in the UC model at the electricity price cap in ERCOT of $4,500 per MWh. As such, NSE essentially would be "dispatched" by the model in hours when the cost of electricity would exceed this cap. Very little NSE occurs over the entire summer at any price. The maximum quantity of NSE for a scenario is 8 MWh over the entire summer, a negligible fraction of total electricity demand. Indeed, the observed NSE may not even represent electricity that would go unserved, but rather could be an artifact of the solution algorithm used in the UC model. Overall, the negligible amount of NSE indicates that sufficient flexibility in the grid exists to accommodate time-differentiated prices while still meeting demand.

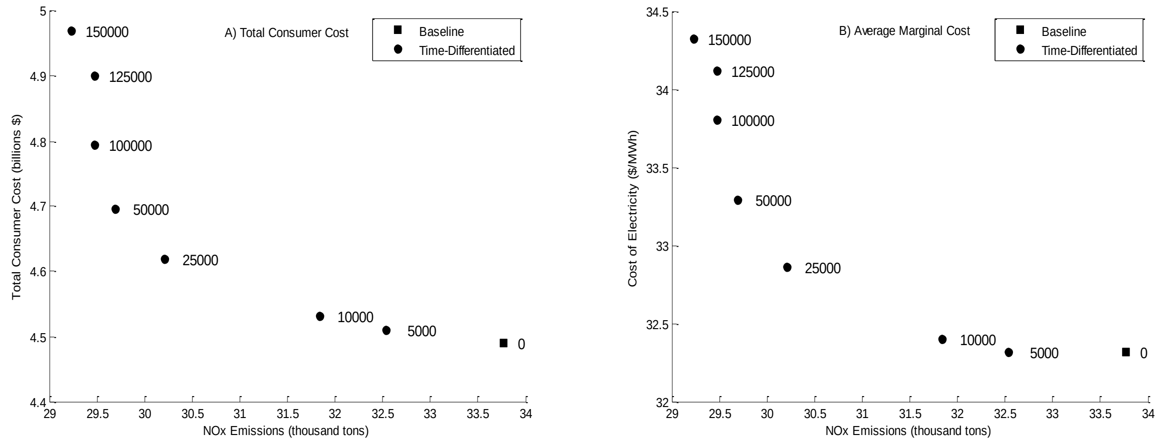
7.1.3: Costs to Consumers

The previous two sections have addressed producer costs, which are an important component of economic losses incurred from policy implementation. In this section, costs to consumers are presented, which are also a crucial policy consideration. Figure 16 (left) shows costs to consumers over the entire summer under time-differentiated prices. These costs exhibit the same trend as costs to consumers on high ozone days. Over both periods of time, costs to consumers rise significantly at high time-differentiated prices, but increase only slightly at low prices. Over the entire summer, for example, consumer costs increase by 10% at $150,000 per ton, but by less than 1% at $5,000 per ton. Nearly all of these additional costs to consumers ($0.4 of the $0.45 billion at $150,000, for example) occur on high ozone days.

Costs to consumers increase by more than costs to producers under time-differentiated pricing. For example, over the entire summer, producer costs increase by just 5% at $150,000 per ton, roughly half the increase in consumer costs. This occurs because electricity generators are remunerated based on the operating cost of the marginal unit in each hour, not their own operating cost. Consequently, as the cost of the marginal unit increases – as it does under time-differentiated pricing (Figure 16, right) – this translates to an increased cost to consumers for all electricity generated, even from nuclear plants or renewables with zero NOx emissions. When calculating producer costs, however, the cost of electricity from such zero-NOx-emitting plants would not increase.

The divergence in consumer and producer costs underscores the necessity to consider both. While a policy may only lead to small losses in the economy in the aggregate (e.g., impose small producer costs), that same policy may be politically untenable if it imposes large additional costs on consumers. Many equity considerations also accompany the issue of increased costs to consumers, like the disproportionate effect on lower income households.

Figure 16: Cost to consumers (A) and the average cost of electricity (B) versus NOx emissions over the entire summer in ERCOT under various time-differentiated prices (circles, labeled in $/ton).



7.2: Time-Differentiated Pricing in PJM

7.2.1: Emissions and Costs on High Ozone Days

As in ERCOT, higher time-differentiated prices in PJM reduce greater amounts of NOx emissions but at a greater average and marginal cost per ton of NOx reduced. NOx emission reductions in PJM range from 20% at $5,000 per ton to 75% at $150,000 per ton (Figure 17), but the average costs of these reductions decreases by a factor of six over the same price range (Table 7).

PJM differs from ERCOT in several ways. First, emission reductions and cost increases at each price in PJM are greater relative to baseline than those in ERCOT for the same price. For instance, at $5,000 per ton, emissions have a smaller decrease in ERCOT (15%) than in PJM (20%), but costs increase more in PJM (2%) than in ERCOT (0.4%). This also holds true at high prices – at $150,000 per ton, emissions decrease less in ERCOT (50%) than in PJM (75%), but costs increase more in PJM (37%) than in ERCOT (13%). At the same time, though, the average cost of emissions reductions in PJM is lower than in ERCOT (Table 7).

PJM and ERCOT also differ in that PJM emissions start at a much higher baseline and decrease to a much lower level than in ERCOT. Additionally, whereas in ERCOT each increase in price achieved more emission reductions, the same is not true in PJM – negligible emission reductions are gained by increasing the price from $125,000 to $150,000 per ton.

Figure 17: Total cost versus NOx emissions on high ozone days in PJM under various time-differentiated prices (circles, labeled in $/ton).

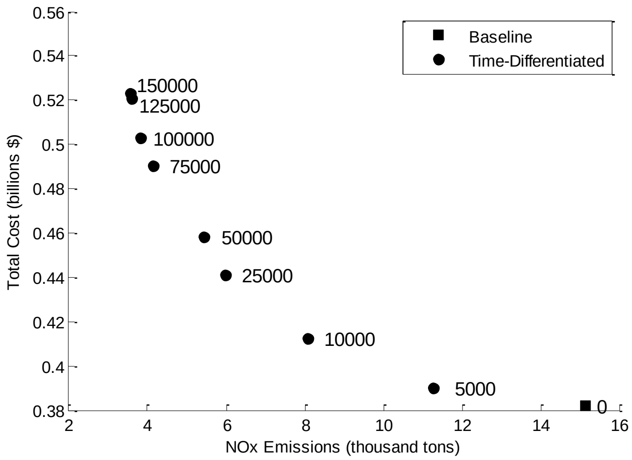


Table 7: Cost and emissions results on high ozone days in PJM under various time-differentiated prices.

Time-Differentiated NOx Price (thousand $/ton)	NOx Emissions (thousand tons)	Cost (million $)	NOx Emissions Reduced (thousand tons)	Cost Increase (million $)	Average Cost per Ton NOx Reduced (thousand $/ton)
0	15.13	381.89	0.00	0.00	N/A
5	11.28	389.98	3.85	8.09	2.10
10	8.08	412.11	7.05	30.22	4.28
25	5.99	440.83	9.14	58.94	6.44
50	5.45	458.35	9.68	76.46	7.90
75	4.15	490.56	10.98	108.67	9.90
100	3.83	503.10	11.30	121.21	10.73
125	3.58	520.47	11.55	138.58	12.00
150	3.56	523.14	11.57	141.24	12.21

Despite these differences, time-differentiated pricing has a similar effect on dispatching in PJM as in ERCOT. Coal-fired generation decreases by as much as 50% under time-differentiated prices explored here relative to no price, and that electricity is provided instead by gas-fired generators (Figure 18). This decrease in coal-fired generation in PJM is similar to that in ERCOT under time-differentiated pricing in percentage and absolute terms. For instance, at a $150,000 per ton price, about 5 terawatt-hours (TWh) of coal-fired electricity is eliminated under time-differentiated pricing in both systems. Also as in ERCOT, total costs in PJM are largely energy costs (Table 8). Fixed costs compose a small fraction of total costs and only at the higher time-differentiated prices explored here when SCR is installed, and start-up costs make up a small and mostly fixed percent of total costs.

Figure 18: Percent change in electricity generation from baseline under various time-differentiated prices (labeled in thousand $/ton) in PJM on high ozone days at coal- (black) and gas- (gray) fired generators.

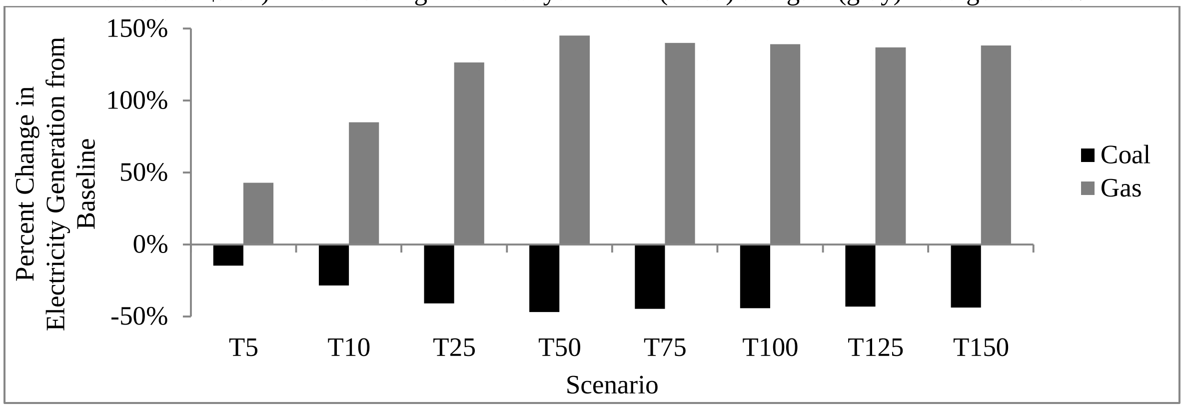


Table 8: Total and disaggregated costs under time-differentiated prices in PJM on high ozone days.

Time-Differentiated NOx Price (thousand $/ton)	Total Cost (million $)	Energy Cost (million $)	SCR Fixed Cost (million $)	Start-up Cost (million $)
0	381.89	364.56	0.00	17.30
5	389.98	373.19	0.00	16.79
10	412.11	394.75	0.00	17.35
25	440.83	422.77	0.00	18.06
50	458.36	439.92	0.00	18.44
75	490.56	457.18	15.35	18.03
100	503.10	464.44	20.62	18.04
125	520.47	471.63	30.85	17.99
150	523.14	474.14	30.85	18.14

The fact that time-differentiated pricing reduces NOx emissions more in PJM than in ERCOT is attributable to two factors: the average NOx emissions rate among the coal fleet in each system and SCR installations. The same amount of redispatching of gas for coal occurs in both systems, so that does not explain the larger reduction in PJM in and of itself. But this fact combined with the higher average NOx emissions rate of coal-fired generators in PJM than in ERCOT means that reduced coal-fired generation in PJM yields greater NOx emission reductions. Coal-fired generators in ERCOT emit an average of 1.3 pounds of NOx per MWh of electricity generated, whereas they emit over twice that, or 2.9 pounds per MWh, in PJM. More SCR installations in PJM than in ERCOT – specifically, 3 GW more – also contributes to greater NOx emission reductions in PJM; the more SCR that is installed, the more electricity will be generated from coal at reduced NOx emissions rates.

The higher average NOx emissions rate at coal-fired generators in PJM also explains why NOx emission reductions are less costly on average in PJM. At higher prices when SCR is triggered, each SCR installed yields greater NOx emission reductions than from plants in ERCOT because of the minimum floor to which SCR can reduce NOx emissions rates. Thus, the fixed costs incurred from SCR achieve greater NOx emission reductions in PJM. But even at lower prices that do not incentivize SCR adoption, each unit of electricity displaced from coal-fired generation in PJM yields on average more than twice the NOx emission reductions from the same unit of electricity displaced in ERCOT. This is partly counterbalanced by the fact that coal-fired generators in PJM are the cheapest generators unlike in ERCOT, so the loss of their electricity generation will increase costs more in PJM. The net effect is that the greater emission reductions obtained outweigh greater costs, resulting in more cost-effective emission reductions in PJM.

The third difference between ERCOT and PJM – that emissions and costs do not appreciably change between $125,000 and $150,000 per ton prices only in PJM – is likely due to saturation of SCR installations at coal facilities at $125,000. No additional plants install SCR at $150,000, so the dispatch situation does not really change (Figure 17). The same coal-fired generators operating at $125,000 operate at $150,000, and gas and nuclear fill in the rest; no additional coal plants are sufficiently incentivized within this mix to also install SCR, which would further drive reductions in emissions. Conversely, more coal plants do install SCR in ERCOT at $150,000 than at $125,000 per ton prices. The fact that no additional emission reductions are achieved in PJM at $150,000 also suggests that little to no additional gas-fired capacity exists to displace coal-fired generation.

7.2.2: Emissions and Costs for Whole Summer

As in ERCOT, summer-wide NOx emission reductions under time-differentiated pricing are slightly *lower* than reductions on high ozone days, likely for the same reasons. Over the entire summer, time-differentiated prices tested here reduce NOx emissions between 8% and 24% (Figure 19). These quantities are substantial, but are achieved at steeply-increasing average cost per ton of NOx emissions reduced (Table 9); costs particularly increase at the $75,000 per ton price, when plants begin to install SCR and incur fixed costs (Table 10) at no benefit for emissions outside of high ozone days (since no financial incentive exists to operate SCR on non-high ozone days). Indeed, beyond $75,000 per ton, summer-wide energy costs remain largely flat, and increases in total costs are mostly driven by increasing fixed costs as more plants install SCR.

Figure 19: Cost versus NOx emissions over the entire summer in PJM under various time-differentiated prices (labeled in $ per ton).

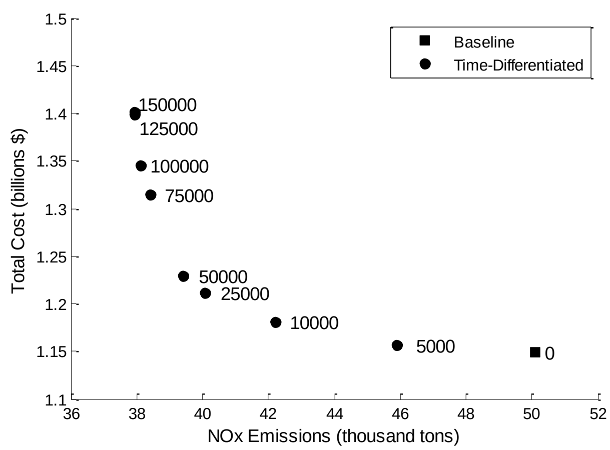


Table 9: Cost and emissions over the entire summer in PJM under time-differentiated prices.

Time-Differentiated NOx Price (thousand $/ton)	NOx Emissions (thousand tons)	Cost (million $)	NOx Emissions Reduced (thousand tons)	Cost Increase (million $)	Average Cost per Ton NOx Reduced (thousand $/ton)
0	50.11	1,148.60	0.00	0.00	N/A
5	45.90	1,156.86	4.21	8.26	1.96
10	42.20	1,180.52	7.91	31.92	4.03
25	40.09	1,210.99	10.02	62.39	6.23
50	39.43	1,229.16	10.68	80.56	7.54
75	38.41	1,314.52	11.70	165.92	14.18
100	38.11	1,345.46	12.00	196.86	16.41
125	37.93	1,398.86	12.17	250.26	20.56
150	37.93	1,401.60	12.18	253.00	20.78

Table 10: Total and disaggregated costs under time-differentiated prices in PJM over the entire summer.

Time-Differentiated NOx Price (thousand $/ton)	Total Cost (billion $)	Energy Cost (billion $)	SCR Fixed Cost (billion $)	Start-up Cost (billion $)
0	1.149	1.095	0	0.054
5	1.157	1.104	0	0.053
10	1.181	1.126	0	0.055
25	1.211	1.154	0	0.057
50	1.229	1.171	0	0.058
75	1.315	1.188	0.069	0.057
100	1.345	1.196	0.093	0.057
125	1.399	1.203	0.139	0.057
150	1.402	1.206	0.139	0.057

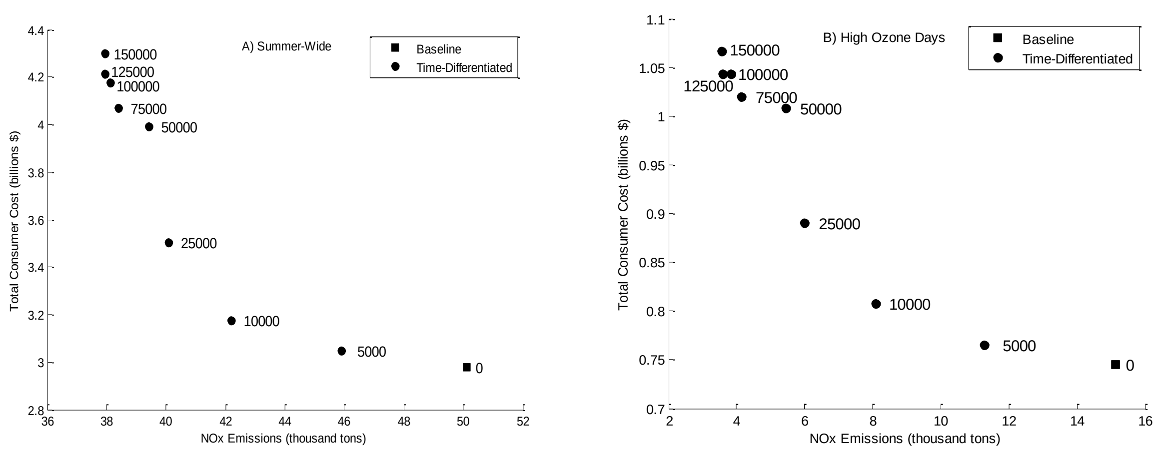
Also similarly to ERCOT, very little non-served energy results during the entire summer across all simulated prices. The maximum amount of NSE in any run is roughly 33 MWh, a tiny fraction of total electricity generation over the course of the year. The insignificant quantity of NSE observed across all prices indicates that sufficient flexibility in PJM exists even at the highest time-differentiated prices tested to meet demand without hitting the offer cap of $1,000/MWh in PJM, to which the cost of NSE is set.

7.2.3: Costs to Consumers

Costs to consumers averaged over the summer rise severely at the highest time-differentiated prices in PJM, on the order of 43% at $150,000 per ton (Figure 20). Cost increases are more moderate at lower prices, around 3% at $5,000, for instance. These increases are much larger than in ERCOT, where costs to consumers rose by just 10% and less than 1% at $150,000 and $5,000 per ton, respectively. Surprisingly, increases in consumer costs on high ozone days in PJM account for less than half of the increases in summer-wide consumer costs. For instance, at $50,000 and above in PJM, costs on high ozone days increase by about $0.25 billion, but summer-wide costs increase by $1 to $1.3 billion. Conversely, in ERCOT increased costs on high ozone days and summer-wide were nearly equal.

The same factor that explained greater producer costs in PJM than in ERCOT also explains greater consumer costs in PJM: where coal-fired generation falls in the merit order in each system. In PJM, coal is the lowest cost generator, so not dispatching coal-fired generators (as may occur under time-differentiated pricing) can significantly increase the electricity price (i.e., the operating cost of the marginal generator), which drives up the costs to consumers. In ERCOT, costs rise less when coal-fired generation is not dispatched because of a higher share of low cost gas-fired generation. Thus, costs to consumers are expected to increase more in PJM than in ERCOT because of greater increases in electricity prices. Indeed, the average marginal cost of electricity over the entire summer increases much more in PJM (25% at $75,000 and above) than in ERCOT (10% at $150,000). Elevated electricity prices may also extend beyond high ozone days in both systems because of operating constraints and start-up costs of coal-fired generators. The increase in prices on non-high ozone days, though, would be greater in PJM than in ERCOT for the above reasons, which could account for high consumer costs incurred on non-high ozone days in PJM but not ERCOT.

Figure 20: Consumer costs versus NOx emissions in PJM over the entire summer (A) and on high ozone days (B) for time-differentiated prices.



7.3: Summary and Discussion

Time-differentiated pricing offers significant reductions in NOx emissions on high ozone days – and possibly the frequency of ozone exceedances – across different types of power systems. Here, results from PJM and ERCOT are presented, which represent coal- and gas-dominated systems, respectively. Alternatively, these systems can be thought of as representing "high-NOx" and "low-NOx" systems, since the majority of generation in PJM is by high-NOx-emitting generators whereas in ERCOT it is by low-NOx-emitting generators. Most U.S. power systems can be roughly thought of as falling into one of these two categories, suggesting that the findings here are broadly applicable to power systems across the U.S.

Results presented here show that the time-differentiated price assessed on NOx emissions can be tailored to achieve a desired reduction in NOx emissions. Emissions and costs are highly responsive to the time-differentiated price. Over the prices studied in ERCOT, emission reductions range from 15% to 50% and costs range between 1% and 13%. Consequently, time-differentiated pricing can be used to achieve moderate emissions reductions at very little cost by selecting a low time-differentiated price. This is true regarding both producer and consumer costs – at a $5,000 per ton time-differentiated price, for instance, costs per consumer per year in ERCOT and PJM would rise by just $1 and $3, respectively. Higher emissions reductions can also be achieved, but at greater total cost and average cost per ton of NOx emissions reduced.

Emissions reductions obtained through time-differentiated pricing primarily stem through substitution of gas- for coal-fired generation. Indeed, at modest prices, no plants installed SCR in PJM or ERCOT, meaning all changes in emissions and costs at these prices result from redispatching. At higher prices when SCR is installed, though, additional emissions and costs are derived largely from the cost and operation of these installations.

Because substitution is the primary driver of costs and emissions, the impacts of time-differentiated pricing are shown to vary depending on the characteristics of the power system. In systems that rely more on coal-fired generation with higher NOx emissions rates (e.g., PJM), time-differentiated pricing offers greater emission reductions on high ozone days. For the results shown, this strategy is more cost-effective for PJM on high ozone days. Whether this holds true in other "high-NOx" systems will depend on plants' average NOx emissions rates and on the position of coal plants within the merit order.

Importantly, even at the highest prices tested here, sufficient flexibility on the grid exists to meet demand without incurring non-served energy (i.e., without hitting the price cap in each system). This suggests that time-differentiated pricing could be implemented today in most systems throughout the U.S. without strong concern about harming the reliability of the power system.

Finally, time-differentiated pricing offers no additional benefits for reducing summer-wide NOx emissions beyond reductions obtained on high ozone days. Virtually all summer-wide emission reductions in both systems stemmed from reductions on high ozone days. In fact, in some cases summer-wide emission reductions were less than reductions on high ozone days due to start-up emissions and other factors. Thus, for reducing emissions on non-high ozone days, time-differentiated pricing is an inappropriate policy instrument.

Chapter 8: Comparison to Alternative Regulatory Instruments

In the previous chapter, the effects of time-differentiated pricing on emissions and costs were presented. These results suggested that time-differentiated pricing would be an effective strategy for reducing emissions on high ozone days – and potentially for reducing the frequency of ozone exceedances – but not across other summer days. Yet, time-differentiated pricing is not the only regulatory approach that has been or will be considered. Comparing time-differentiated pricing to other regulatory strategies is therefore crucial to understanding its merits.

In this chapter, time-differentiated pricing is compared to two alternative regulatory instruments – undifferentiated NOx pricing and technology-based standards. Both regulatory instruments are comparable to current approaches for regulating NOx emissions across the U.S. Technology-based standards currently limit NOx emissions at new and existing sources throughout the nation (see Section 3.1.1). Undifferentiated pricing, although it has not been directly implemented by any state, is comparable to cap-and-trade programs that have been implemented in the eastern half of the U.S., including in Texas, and California (see Section 3.1.2). These markets assign a cost per unit of emissions through allocated emission permits. Moreover, existing cap-and-trade programs do not differentiate NOx emissions within the summer, like the undifferentiated price modeled here. The formulation for each regulatory scheme is provided in Section 5.3 above.

As in the previous chapter, emissions and costs are presented for just high ozone days as well as for the entire summer. While summer-wide costs encapsulate high ozone day costs and therefore are more representative of the actual total costs of a policy, variability in the number of high ozone days over a summer will affect summer-wide costs. Considering costs only on high ozone days therefore yields important insights into the costs of policies. As in the previous chapter, the costs presented below are producer costs, unless explicitly indicated as consumer costs.

8.1: Undifferentiated Pricing

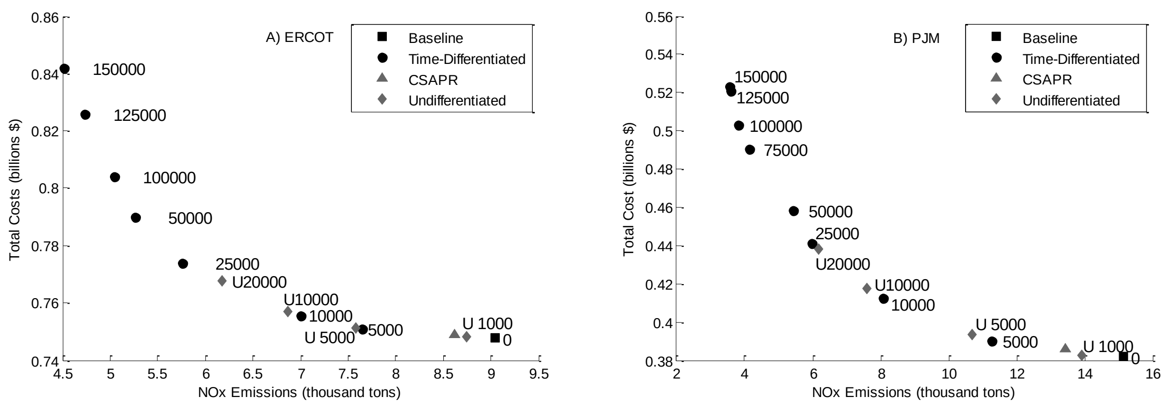
One alternative to time-differentiated pricing for reducing NOx emissions on high ozone days is an undifferentiated price that imposes the same price for NOx emissions throughout the entire summer, regardless of when NOx emissions occur. An undifferentiated price is similar to the current cap-and-trade Clean Air Interstate Rule (CAIR) in that both treat all NOx emissions in the ozone season (i.e., the summer) equivalently. Undifferentiated prices are also similar to cap-and-trade programs, as the latter effectively assigns a cost on emissions through the price of emissions permits issued under the program. In theory, aggregate emissions and costs should be identical under the two regulatory schemes if the emissions permit price is equal to the undifferentiated price.

Undifferentiated NOx prices of $1,000, $5,000, $10,000 and $20,000 per ton are considered. An undifferentiated price of $500 per ton on NOx and SO_2 emissions is used to simulate the likely outcomes under the Cross-State Air Pollution Rule (CSAPR), the proposed successor to the CAIR. Emissions budgets under the CSAPR were set such that NOx and SO_2 emissions permits would be priced at roughly $500 per ton (U.S. Environmental Protection Agency, 2011b). Prices above $20,000 are not considered because such high prices are unlikely to be politically feasible in the near-term, given prices in comparable past and proposed regulations (such as the CSAPR).

8.1.1: Emissions and Costs on High Ozone Days

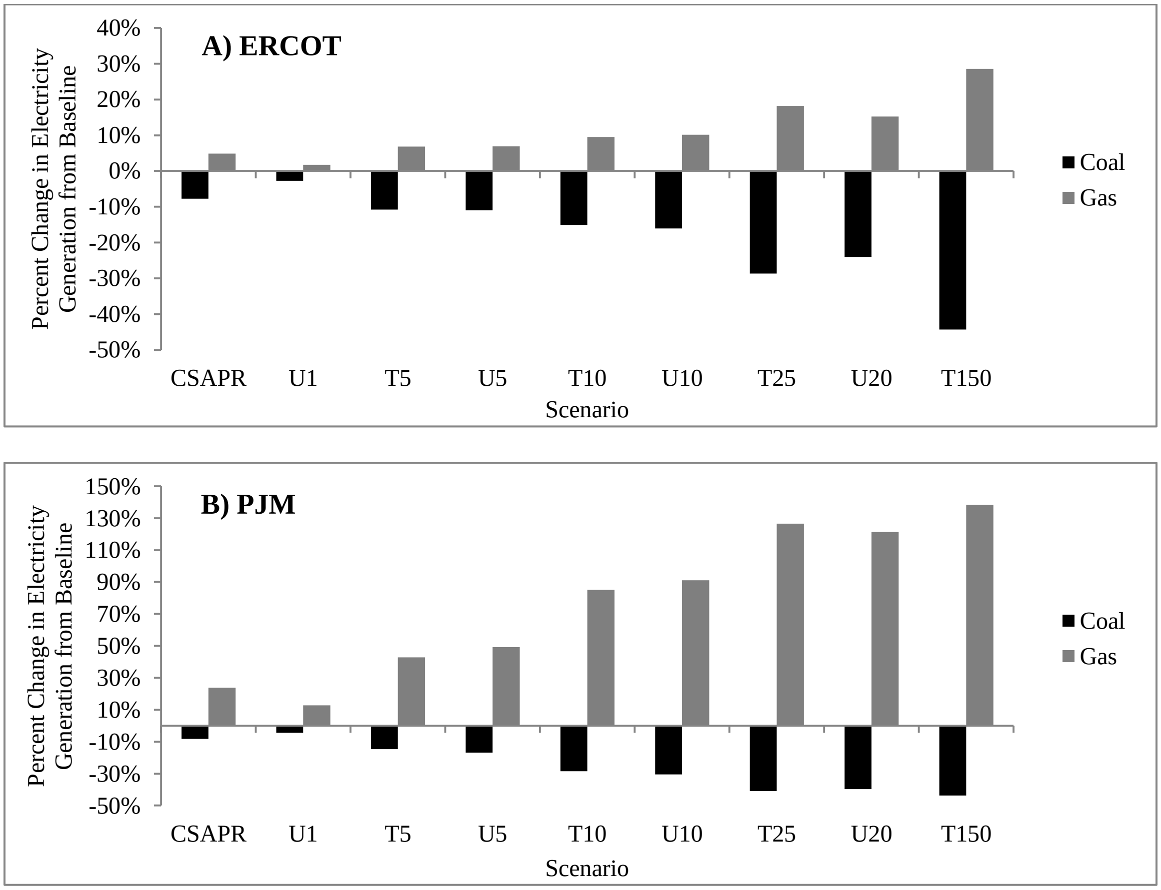
Undifferentiated pricing almost exactly reproduces emissions and costs on high ozone days that are achieved under time-differentiated pricing in both PJM and ERCOT (Figure 21). Emission reductions are slightly greater at undifferentiated prices than at the same time-differentiated prices, but costs are also slightly greater. Consequently, the two sets of points (undifferentiated and time-differentiated prices) trace a very similar curve in the emissions-cost space, suggesting the two types of policies are equivalent with regard to their impacts on emissions and costs high ozone days.

Figure 21: Costs versus NOx emissions on high ozone days in ERCOT (A) and PJM (B) for various time-differentiated prices (black circles, labeled with the price in $/ton) and un-differentiated prices (gray diamonds, labeled with 'U' and the price in $/ton). An undifferentiated price based on outcomes expected under the CSAPR (gray triangle) is also presented.



The outcomes are similar on high ozone days under the two pricing systems because they similarly affect dispatching on high ozone days within PJM and ERCOT. No SCR installations are triggered under any of the undifferentiated price scenarios, so emission and cost impacts from undifferentiated pricing stem primarily from substitution of gas- for coal-fired generation. This is also true for comparable time-differentiated prices. Moreover, in PJM and ERCOT comparable time-differentiated and undifferentiated prices lead to similar levels of substitution of gas- for coal-fired generation (Figure 22). For instance, coal- and gas-fired generators produce about the same amount of electricity (9 and 5.5 TWh, respectively) on high ozone days under a $5,000 per ton time-differentiated price as under an undifferentiated price in ERCOT. Given that energy costs make up the vast majority of total costs, similar patterns of redispatching for time-differentiated and undifferentiated pricing also explains why costs are similar under comparable time-differentiated and undifferentiated prices.

Figure 22: Percent change in electricity generation from baseline on high ozone days by coal-fired (black) and gas-fired (gray) generators under time-differentiated prices (labeled with 'T' and the price in thousands $/ton) and undifferentiated prices (labeled with 'U' and the price in thousands $/ton). An undifferentiated price based on outcomes expected under the CSAPR is also presented. Results are presented for ERCOT (A) and PJM (B).

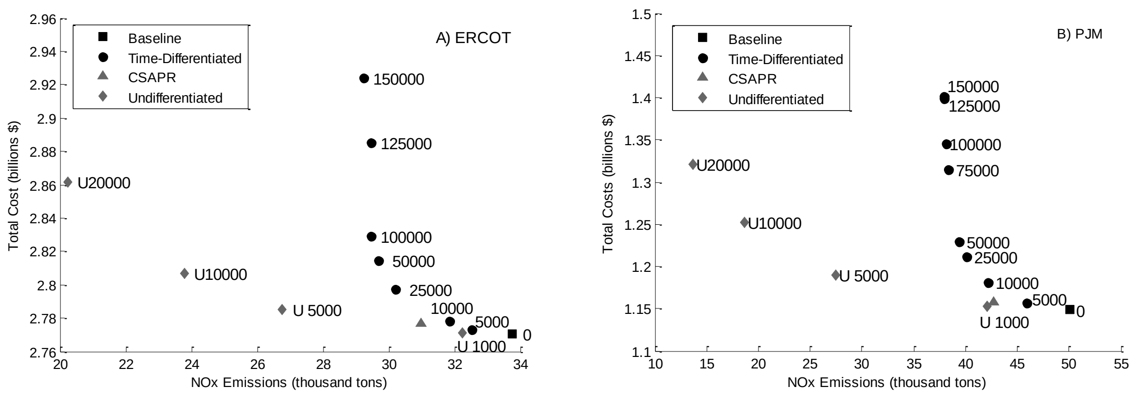


8.1.2: Emissions and Costs for Whole Summer

Unlike on high ozone days, emissions and costs over the entire summer are very different under time-differentiated versus undifferentiated pricing. In PJM and ERCOT, undifferentiated pricing produces significantly greater emission reductions over the course of the summer than comparable time-differentiated regulations (Figure 23). For instance, a $5,000 per ton undifferentiated price in PJM reduces NOx emissions over the entire summer by nearly 50%, over four times more than a $5,000 per ton time-differentiated price. In fact, undifferentiated prices of $5,000 per ton or more yield greater emission reductions over the entire summer in PJM and ERCOT than the highest time-differentiated price of $150,000 per ton. This finding is not surprising because a price signal over the entire summer would be expected to reduce NOx emissions over the entire summer more than a price signal sent on only a small number of summer days.

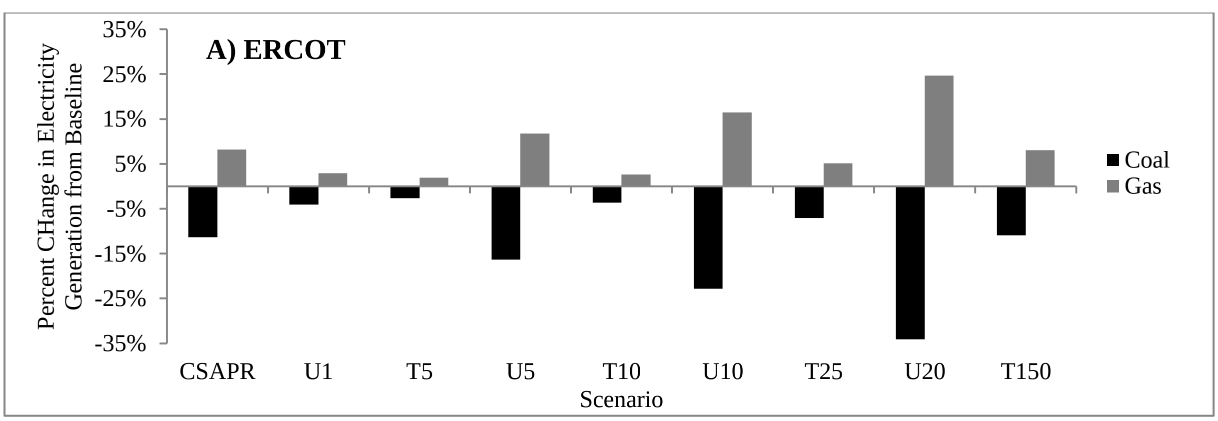
Not only does undifferentiated pricing yield much greater emission reductions over the entire summer, it also does so much more cost-effectively. Summer-wide costs under a $5,000 per ton undifferentiated price, for instance, are less than under time-differentiated prices of $25,000 and greater, despite achieving far greater emission reductions. Consequently, any desired level of summer-wide NOx emission reductions can be achieved at much lower cost under undifferentiated pricing as compared to time-differentiated pricing.

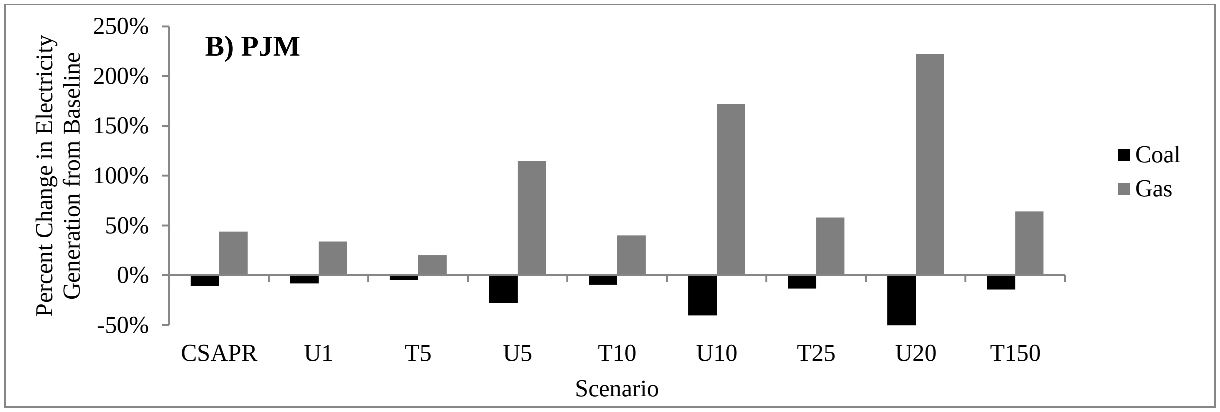
Figure 23: Costs versus NOx emissions for the entire summer in ERCOT (left) and PJM (right) for various time-differentiated prices (circles, labeled in $/ton) and un-differentiated prices (gray diamonds, labeled with 'U' and the price in $/ton). An undifferentiated price based on outcomes expected under the CSAPR (gray triangle) is also presented.



Undifferentiated pricing achieves greater summer-wide emission reductions than time-differentiated prices because it leads to substantially more substitution of gas- for coal-fired generation. Figure 24 shows that in ERCOT, coal-fired electricity generation over the summer falls by about 33% from baseline under an undifferentiated price of $20,000 per ton. This is about 25% below generation levels under even a $150,000 per ton time-differentiated price. There is an even greater difference in coal-fired generation under time-differentiated versus undifferentiated prices in PJM; at a $20,000 undifferentiated price, coal generation is almost 50% below that at a $150,000 time-differentiated price.

Figure 24: Percent change in electricity generation from baseline over the whole summer by coal-fired (black) and gas-fired (gray) generators under time-differentiated prices (labeled with 'T' and the price in thousands $/ton) and undifferentiated prices (labeled with 'U' and the price in thousands $/ton). An undifferentiated price based on outcomes expected under the CSAPR is also presented. Results are presented for ERCOT (A) and PJM (B).





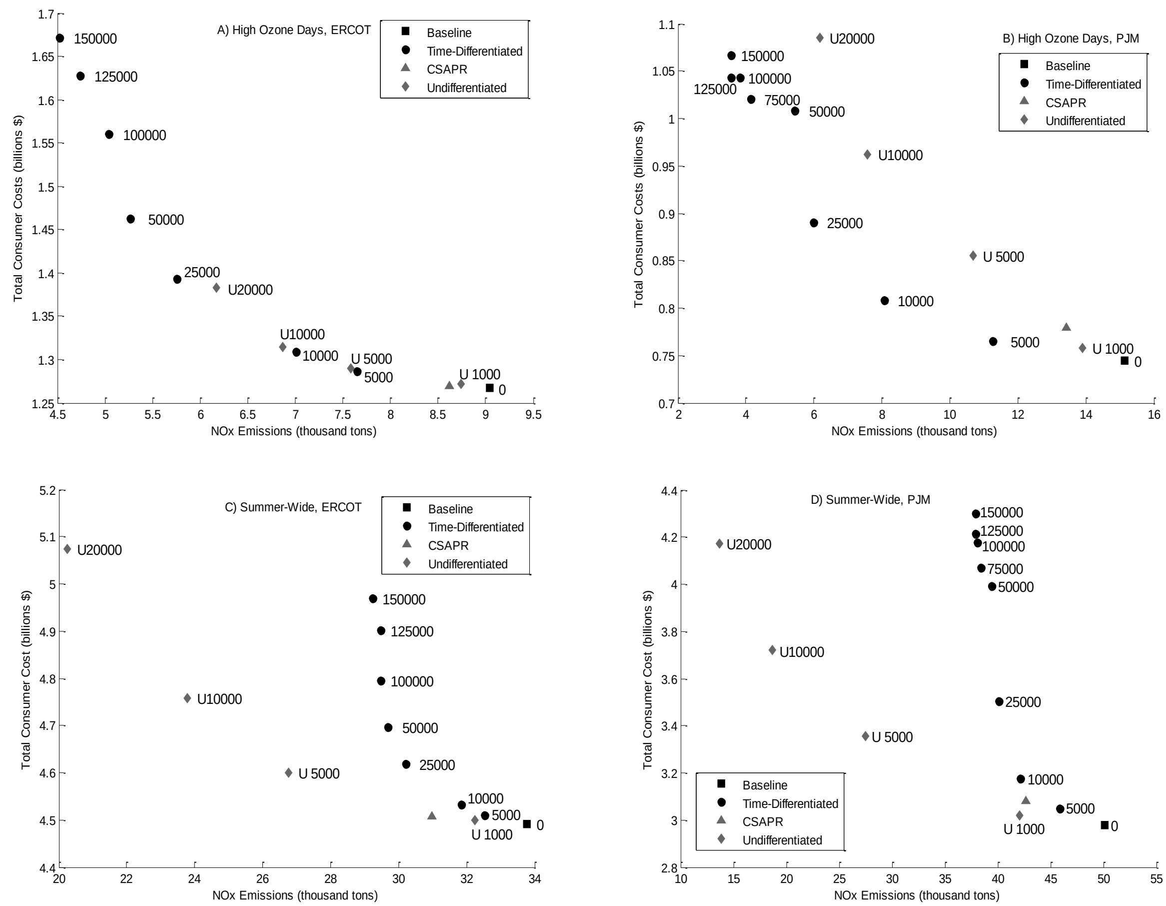
Trivial amounts of non-served energy occur over the entire summer under any undifferentiated prices tested here in ERCOT and PJM, indicating that virtually all electricity demand would continue to be met under the undifferentiated prices.

8.1.3: Costs to Consumers

Costs to consumers on high ozone days are much greater under undifferentiated pricing than time-differentiated pricing in PJM (Figure 25, B). This is in contrast to observations of producer costs in PJM, which are largely the same under the two policies (Figure 21, B). In ERCOT, though, costs to consumers are largely the same between the pricing policies, as are producer costs. In PJM, for instance, a $5,000 per ton undifferentiated price increases costs to consumers by 13% more than a $5,000 per ton time-differentiated price (Figure 25, B). Producer costs under these two policies, though, are largely the same (Figure 21, B).

Cost increases over the entire summer from undifferentiated prices are much greater when measured as consumer rather than producer costs, but the same is true for time-differentiated prices (Figure 25). Thus, undifferentiated pricing is again found to be more cost-effective, this time for consumers, than time-differentiated pricing for reducing summer-wide NOx emissions.

Figure 25: Consumer costs versus NOx emissions on high ozone days (top) and over the entire summer (bottom) in ERCOT (left) and PJM (right) for various time-differentiated prices (circles, labeled with the price in \$/ton) and un-differentiated prices (gray diamonds, labeled with ‘U’ for undifferentiated and then the price in \$/ton). An undifferentiated price based on outcomes expected under the CSAPR (gray triangle) is also presented.

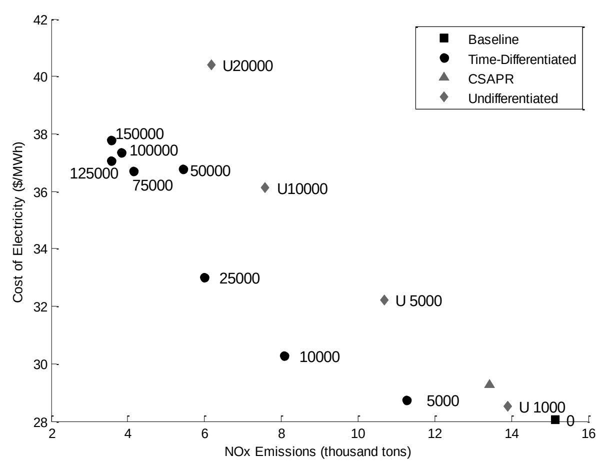


Undifferentiated prices in PJM increase consumer costs much more on high ozone days than time-differentiated prices because undifferentiated prices result in less coal-fired electricity generation on high ozone days. While redispatching of coal-fired generation is similar between comparable time-differentiated and undifferentiated prices in PJM (Figure 22), it is slightly less under the undifferentiated scenario. Coal-fired generation on high ozone days at a \$5,000 time-differentiated price, for instance, is 3% lower than under a \$5,000 undifferentiated price. This is because it may be less economic to run coal-fired generators on high ozone days under undifferentiated pricing than under time-differentiated pricing. Dispatching is determined here in one week intervals. As such, under time-differentiated pricing, it may be economic to operate a coal-fired generator on a high ozone day even if cheaper units could be operated on that day in order to avoid start-up costs later in the week. On non-high ozone days in the week, the hypothetical coal-fired generator would likely generate cheaper power in PJM than gas-fired generators (Figure 9), so operating it on those days would lead to lower costs. Under undifferentiated pricing, on the other hand, the same price is assessed for every day, so it would be less economic to operate coal-fired generators at any point in the summer (evident in Figure 24), reducing the likelihood they would be dispatched on high ozone days.

Less coal-fired generation leads to greater dispatching of gas-fired generation under undifferentiated prices. This redispatching increases electricity prices (Figure 26) because gas-fired generation is more expensive than coal-fired generation in PJM (Figure 9). Greater electricity prices, in turn, increase costs to consumers for all electricity generated in an hour, since generators are remunerated based on the operating cost of the marginal unit.

In ERCOT, on the other hand, gas-fired generators are cheaper relative to coal-fired generation than in PJM. As such, the economic incentive to operate coal-fired generation on high ozone days when dispatching over the course of a week under time-differentiated pricing is less. Dispatching coal-fired generation on non-high ozone days would not reduce costs as much as PJM because cheap gas-fired generation could generate on those days. Consequently, coal-fired electricity generation under comparable time-differentiated and undifferentiated prices in ERCOT is more similar than in PJM, so consumer costs are largely the same between the two pricing schemes.

Figure 26: Average marginal cost of electricity on high ozone days in PJM under time-differentiated (circles, labeled in \$/ton) and undifferentiated (gray diamond, labeled with 'U' and then the price in \$/ton) pricing. An undifferentiated price based on outcomes expected under the CSAPR (gray triangle) is also presented.



8.1.4: Summary and Discussion

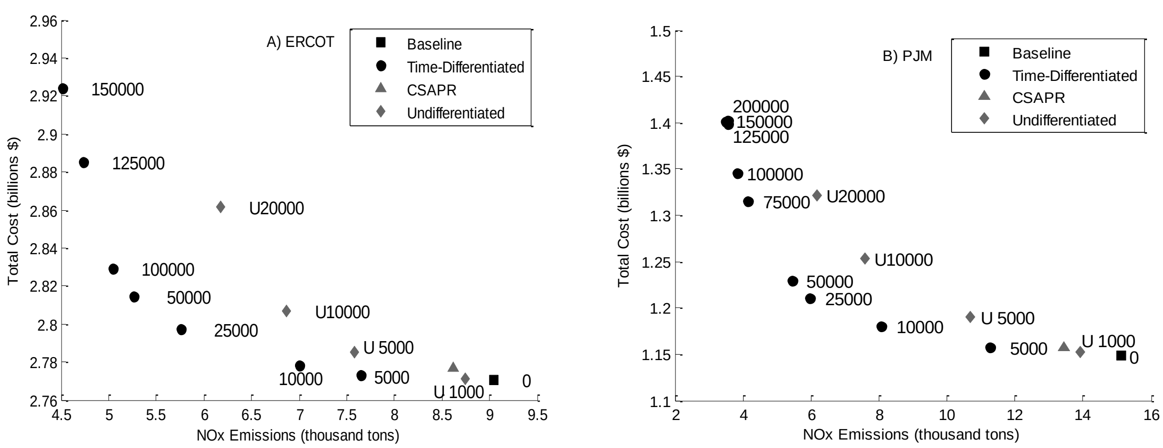
The primary tradeoff between undifferentiated and time-differentiated pricing pertains to when NOx emissions are most desired to be reduced – on high ozone days or over the entire summer. If it is the latter, then undifferentiated pricing is clearly superior. Results presented here show that summer-wide emissions reductions under undifferentiated pricing are far more cost-effective than under time-differentiated pricing in PJM and ERCOT. In other words, to achieve any desired level of summer-wide NOx emissions reductions, undifferentiated pricing has lower costs than time-differentiated pricing. This is not surprising, given that time-differentiated pricing only applies to high ozone days, a small subset of summer days.

If, however, reducing NOx emissions on high ozone days – which, as discussed in the introduction, is still a pressing issue in many areas – is the foremost desired policy outcome, then time-differentiated pricing is the best policy approach for two reasons: it achieves the goal at lower cost and likely leads to fewer coal plant retirements. Each reason is addressed in turn below.

For a given level of NOx emission reductions on high ozone days, a time-differentiated price is less costly than an undifferentiated price on the whole. This is true for producer and consumer costs (Figure 27 shows producer costs). Time-differentiated prices have concentrated costs on high ozone days, whereas undifferentiated prices incur costs over the entire summer. Consequently, in PJM for instance, although a $5,000 undifferentiated price achieves roughly the same emission reductions at roughly the same cost on high ozone days as a $5,000 time-differentiated price, its summer-wide cost is $0.3 billion greater due to costs incurred on non-high ozone days. The increased cost of an undifferentiated price on high ozone days is greater for higher price levels.

Time-differentiated pricing is also less costly than undifferentiated pricing in ERCOT for achieving a desired level of NOx emission reductions on high ozone days. These cost savings are substantial, and are greater for higher price levels on high ozone days (Figure 27). However, cost savings are less than in PJM because of the differences in generation mixes. In systems more reliant on coal-fired generation like PJM, time-differentiated pricing offers greater cost savings relative to undifferentiated pricing than in systems less reliant on coal-fired generation, like ERCOT. One implication of this finding is that as generation mixes become cleaner and more coal plants are retired, cost savings of a time-differentiated price relative to an undifferentiated price will decrease. But even for systems with lower average NOx emissions rates, substantial savings can still be expected from a time-differentiated price for reducing NOx emissions on high ozone days, as observed in ERCOT.

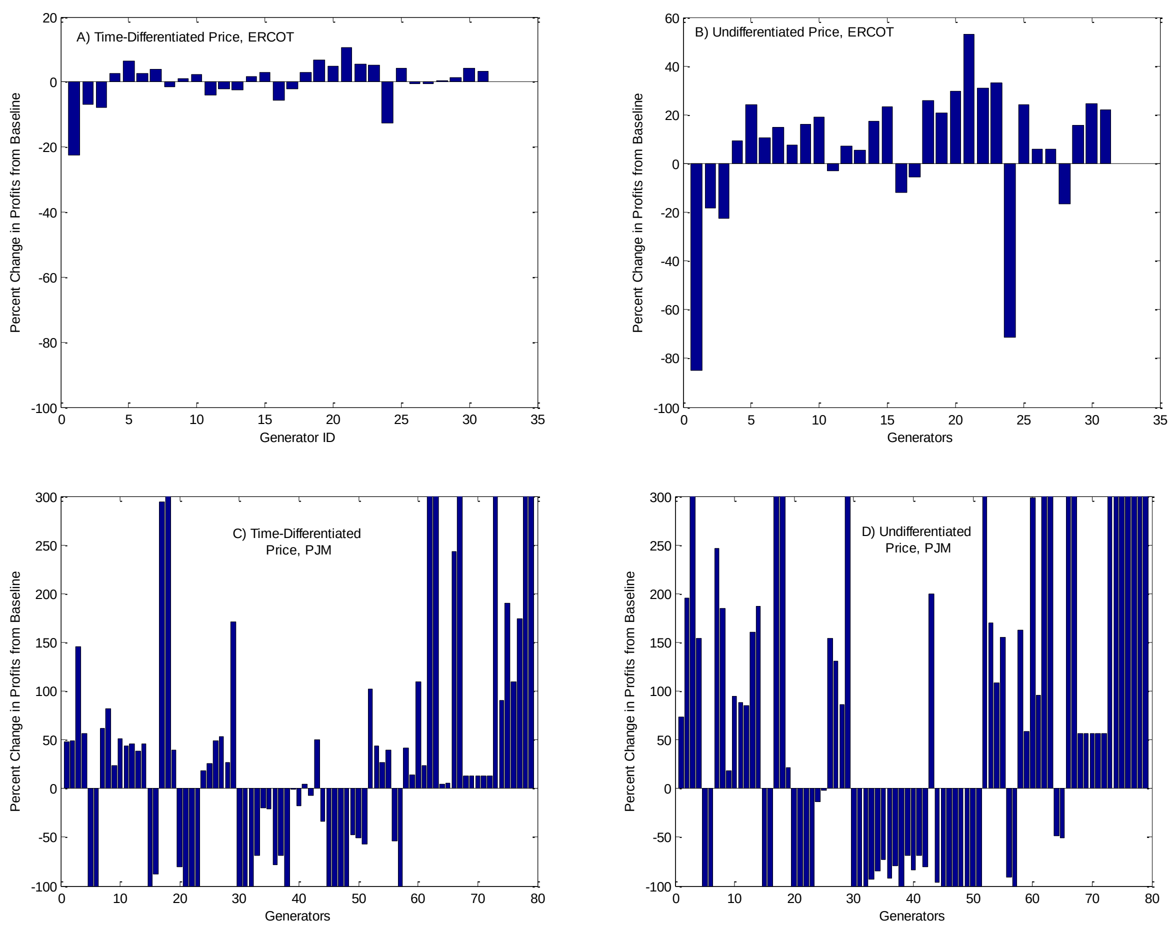
Figure 27: NOx emissions on high ozone days versus summer-wide costs in ERCOT (A) and PJM (B) under various time-differentiated prices (circles, labeled in $/ton) and un-differentiated prices (gray diamonds, labeled with ‘U’ for undifferentiated and then the price in $/ton). An undifferentiated price based on outcomes expected under the CSAPR (gray triangle) is also presented.



The second reason why time-differentiated pricing is a preferred approach for reducing NOx emissions on high ozone days is that it would likely result in fewer coal plant retirements than an undifferentiated price. While the two-phase model used here does not capture coal plant retirements, profits at each coal plant provide some indication of retirements that would be expected under each policy. Retirements are a key consideration in a policy’s viability because they cause acute economic harm to parent companies as well as to nearby towns that benefit from jobs and other economic activity. Companies and towns that would be harmed by plant retirements can form concentrated interest groups, which can seriously undermine the chances of a policy being adopted (Olson, 1965). Thus, a policy that can achieve the same level of emissions while causing fewer plants to retire would have greater odds of adoption.

Time-differentiated pricing would lead to many fewer coal plant retirements than undifferentiated pricing because it redistributes profits among coal plants to a much lesser extent (Figure 28). In ERCOT, although more coal plants have reduced profits under a time-differentiated than undifferentiated price, losses under the latter pricing scheme are much greater in magnitude. Coal generators lose at most 23% under time-differentiated pricing (Figure 28, A), whereas two generators lose over 80% and 60% of their profits under an undifferentiated scheme (Figure 28, B). In PJM, the difference in profits at individual generators under the two pricing regimes is even starker – more plants lose profits and losses are of much greater magnitude under undifferentiated pricing (Figure 28, C and D). Thus, while companies that own coal plants would likely oppose any pricing scheme because they stand to lose profits, this opposition would be much greater to undifferentiated pricing, under which they would stand to lose much more.

Figure 28: Percent change in profits at coal-fired generators under a $10,000 time-differentiated price (left) and $10,000 undifferentiated price (right) relative to baseline (no NOx price) in ERCOT (top) and PJM (bottom).



Coal-fired generators with significant declines in profits under undifferentiated pricing should not necessarily install SCR. Profitability of SCR installation as calculated here includes the annualized capital cost of SCR. For every generator in PJM and ERCOT under an undifferentiated price of even $20,000 per ton, SCR installation – once capital costs were included – incurs negative net revenues (see Section 6.3.3). If SCR installation generated positive profits at any generator, even if total profits at the generator were less than in the baseline scenario without any emissions price, then the two-phase model used here would install SCR at that generator.

8.2: Technology-Based Standards

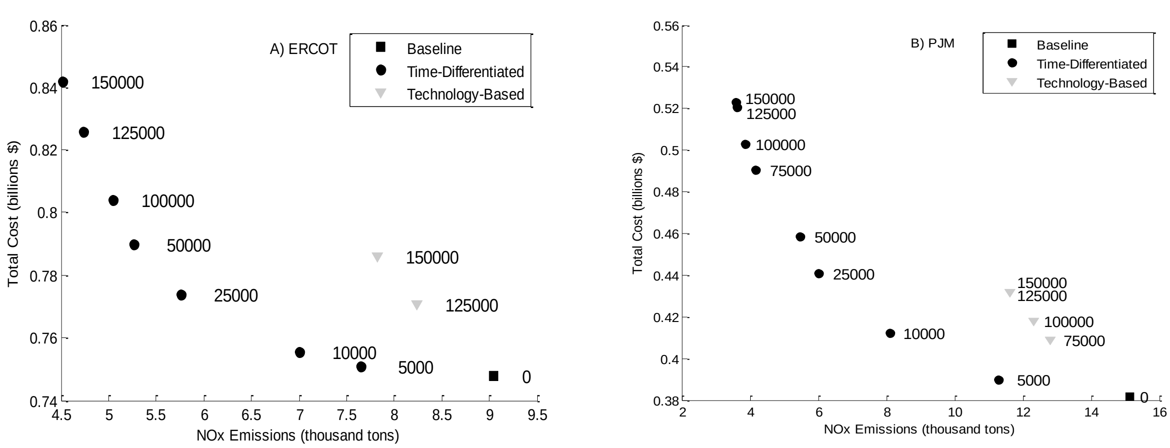
A second alternative policy to compare to time-differentiated pricing is a technology-based standard, under which some plants would be required to install and operate SCR. Technology- and performance-based standards were the primary regulatory approach for reducing NOx emissions from the 1970s through the 1990s. Moreover, if legal challenges to cap-and-trade programs for NOx emissions prevent their implementation (see Section 3.1.2) technology-based standards are a likely alternative.

As discussed in Section 5.3.3, SCR installations mandated under a technology-based standard are allocated among coal-fired generators in two different ways: to those generators that chose to install SCR at the relevant time-differentiated price and to generators at random until the installed capacity of SCR is the same as under the relevant time-differentiated price.

8.2.1: Emissions and Costs on High Ozone Days

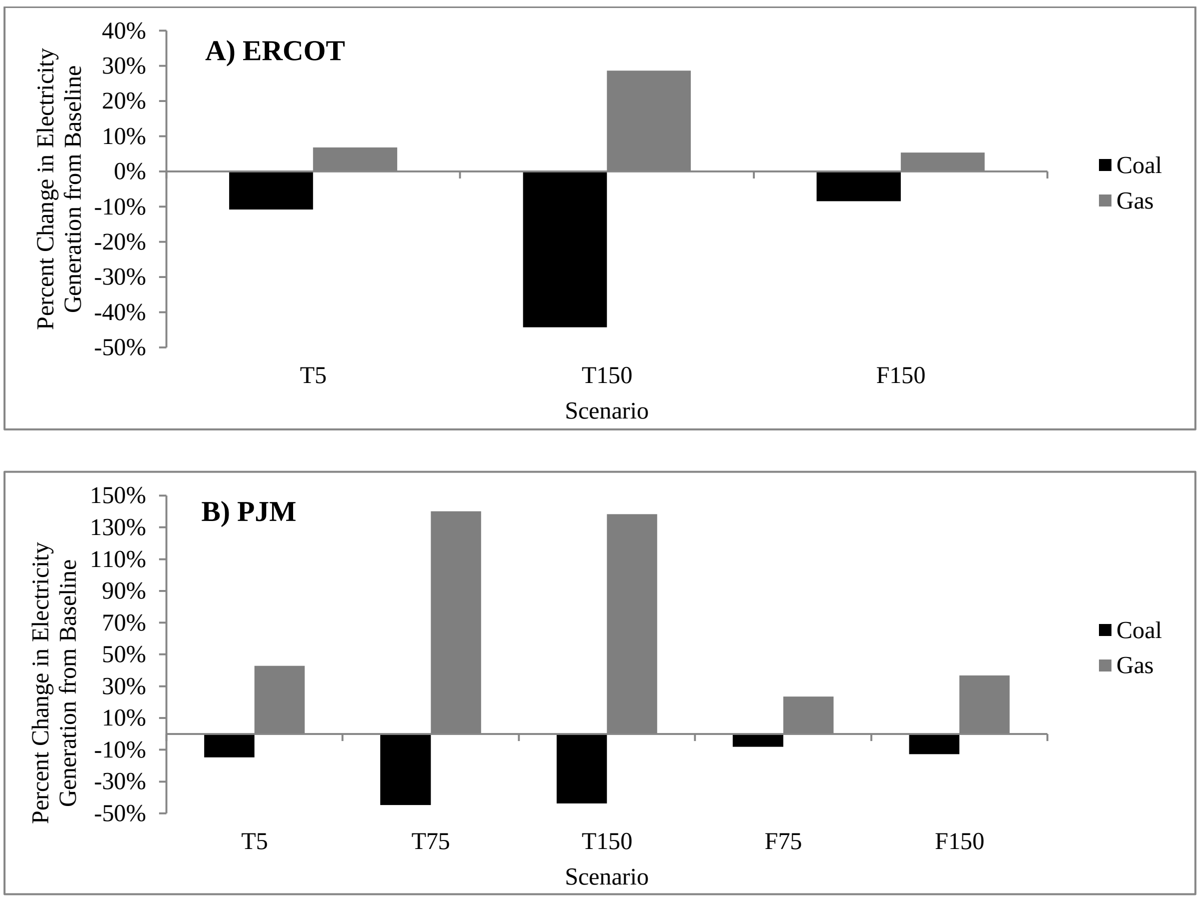
On high ozone days, time-differentiated pricing offers far greater NOx emission reductions and at much lower cost than technology-based standards (Figure 29). In ERCOT and PJM, the lowest time-differentiated price ($5,000 per ton) reduces NOx emissions even more than if SCR was required for all plants that choose to install SCR under the highest time-differentiated price of $150,000 per ton. (At that price, in ERCOT 7 plants with a combined capacity of 4 GW adopt SCR in ERCOT, and in PJM 23 plants with a combined capacity of 7 GW do.) Moreover, greater emissions reductions on high ozone days obtained under time-differentiated pricing are also obtained at lower cost. Thus, any desired level of NOx emission reductions on high ozone days can be achieved at lower cost with time-differentiated pricing than a technology-based standard.

Figure 29: NOx emissions versus costs on high ozone days in PJM (right) and ERCOT (left) under various time-differentiated prices (circles, labeled in $/ton) and technology-based standards (gray triangles, labeled with the time-differentiated price on which the allocated SCR capacity is based).



Time-differentiated pricing offers much greater potential for NOx emission reductions on high ozone days because it reduces NOx emissions from all coal-fired generators. In contrast, technology-based standards only affect NOx emissions at the coal-fired generators that install and operate SCR. Consequently, total coal-fired generation under time-differentiated pricing is much less than under technology-based standards. In PJM, for instance, the lowest time-differentiated price tested ($5,000 per ton) reduces coal-fired generation more than any technology-based standard (Figure 30). This is also true in ERCOT. Less coal-fired generation, in turn, leads to lower NOx emissions. While more coal-fired generation under a technology-based standard may be "cleaner" with respect to NOx emissions because of SCR operation, this does not compensate for the reality that a technology-based standard is unlikely to result in SCR installation at every coal plant for both economic and political reasons.

Figure 30: Percent change in electricity generation from baseline on high ozone days by coal-fired (black) and gas-fired (gray) generators in ERCOT (A) and PJM (B) under time-differentiated prices (labeled with 'T' and the price in thousand $/ton) and technology-based standards (labeled with 'F' and the time-differentiated price on which the allocated capacity of SCR is based).



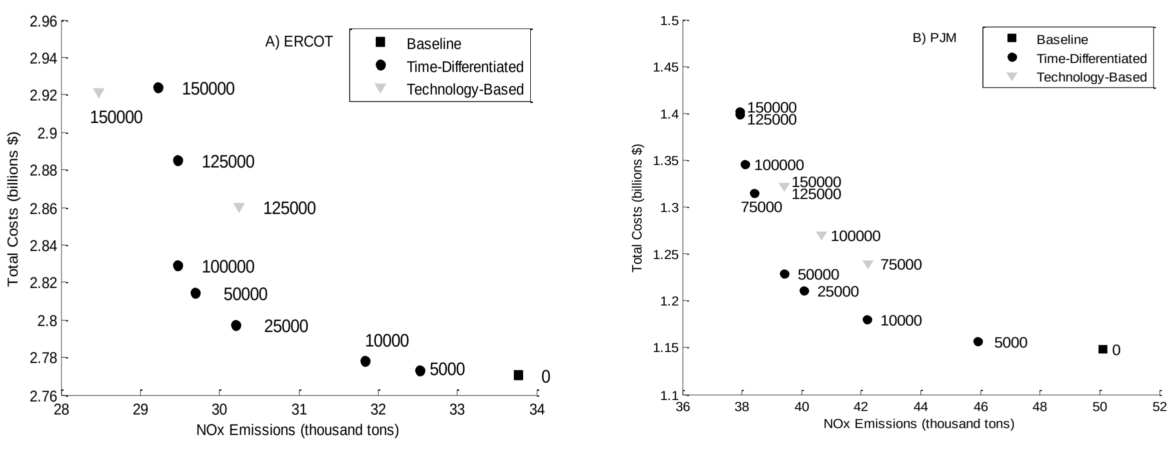
Cost increases relative to baseline under technology-based standards are driven by additional fixed and energy costs. The fixed costs are the capital costs for the SCR installations. The increased energy costs are due to two forces. First, SCR increases operational costs, so generators that install SCR are more expensive to run. Dispatching these generators directly increases energy costs. But because these generators are more expensive to operate, they may be dispatched less than they would have been without SCR. If these plants are dispatched less, then more expensive generators will be dispatched instead, which would also increase energy costs. As before, fixed costs constitute a small portion of total costs across all scenarios.

8.2.2: Emissions and Costs for Whole Summer

Considering aggregate emissions for the entire summer, a more complicated picture emerges regarding the tradeoffs in emission reductions and costs under time-differentiated pricing versus technology-based standards. In PJM, time-differentiated pricing offers a more cost-effective solution for reducing NOx emissions across the entire summer over the range of prices and standards tested here (Figure 31, B). For any given NOx emissions reduction target across the entire summer, time-differentiated pricing can meet that target at lower cost than with a technology-based standard. In ERCOT, on the other hand, technology-based standards tested here offer a more cost-effective method of reducing summer-wide NOx emissions beyond a certain threshold (Figure 31, A).Installing SCR on 7 plants in ERCOT – the same 7 that choose to install SCR under a $150,000 per ton time-differentiated price – achieves greater summer-wide NOx emission reductions than observed under any time-differentiated price here, and at lower cost than a $150,000 time-differentiated price. For lower summer-wide NOx emissions reduction targets (e.g. to around 30 thousand tons), time-differentiated pricing is more cost-effective. Thus, in ERCOT, whether time-differentiated pricing or a technology-based standard reduces summer-wide NOx emissions more cost-effectively depends critically on the desired level of reductions.

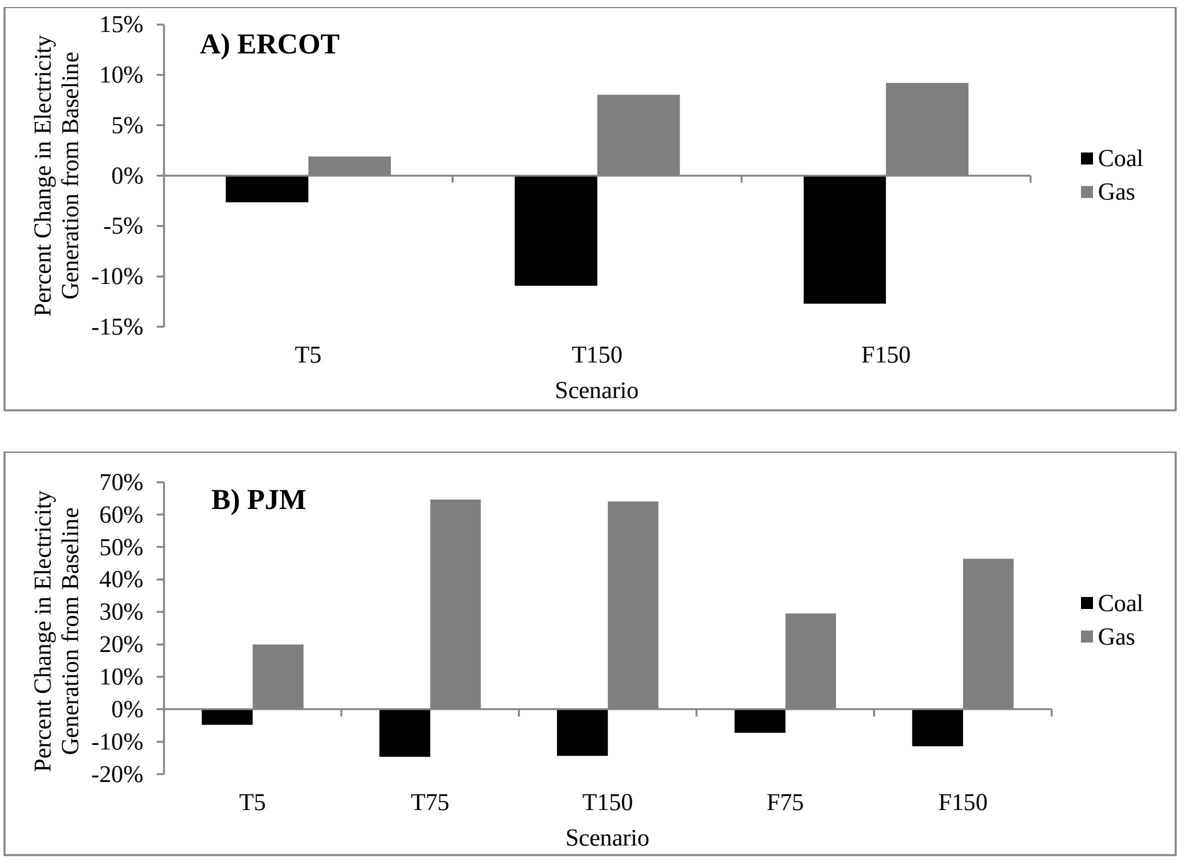
The same conclusion may also be true for PJM, given that the trendline for emissions and costs under technology-based standards in PJM is non-linear but less curved than the trendline under time-differentiated pricing (Figure 31, B). However, that conclusion cannot be made for PJM based on the prices and corresponding technology-based standards explored here.

Figure 31: NOx emissions versus costs for the entire summer in ERCOT (A) and PJM (B) under various time-differentiated prices (circles, labeled with the price in $/ton) and technology-based standards (gray diamonds, labeled with the time-differentiated price on which the installed SCR capacity is based).



As previously discussed, technology-based standards reduce NOx emissions at affected coal plants through two mechanisms, operation of SCR and redispatching of coal plants that become uneconomic with SCR. The relative importance of each mechanism varies between power systems. In PJM, where coal-fired generation is the lowest cost generation, most coal plants with SCR are still economic to operate, so emission reductions stem primarily from SCR operation. This is illustrated by higher coal-fired generation under the technology-based scenarios than in the comparable time-differentiated scenarios in PJM over the entire summer (Figure 32, B). Conversely, in ERCOT where gas-fired generation constitutes a large share of baseload generation, SCR operation at coal plants can make those plants uneconomic. Consequently, summer-wide coal-fired generation in ERCOT is similar under a \$150,000 time-differentiated price (Figure 32, A, 'T150') and a comparable technology-based standard (Figure 32, A, 'F150').

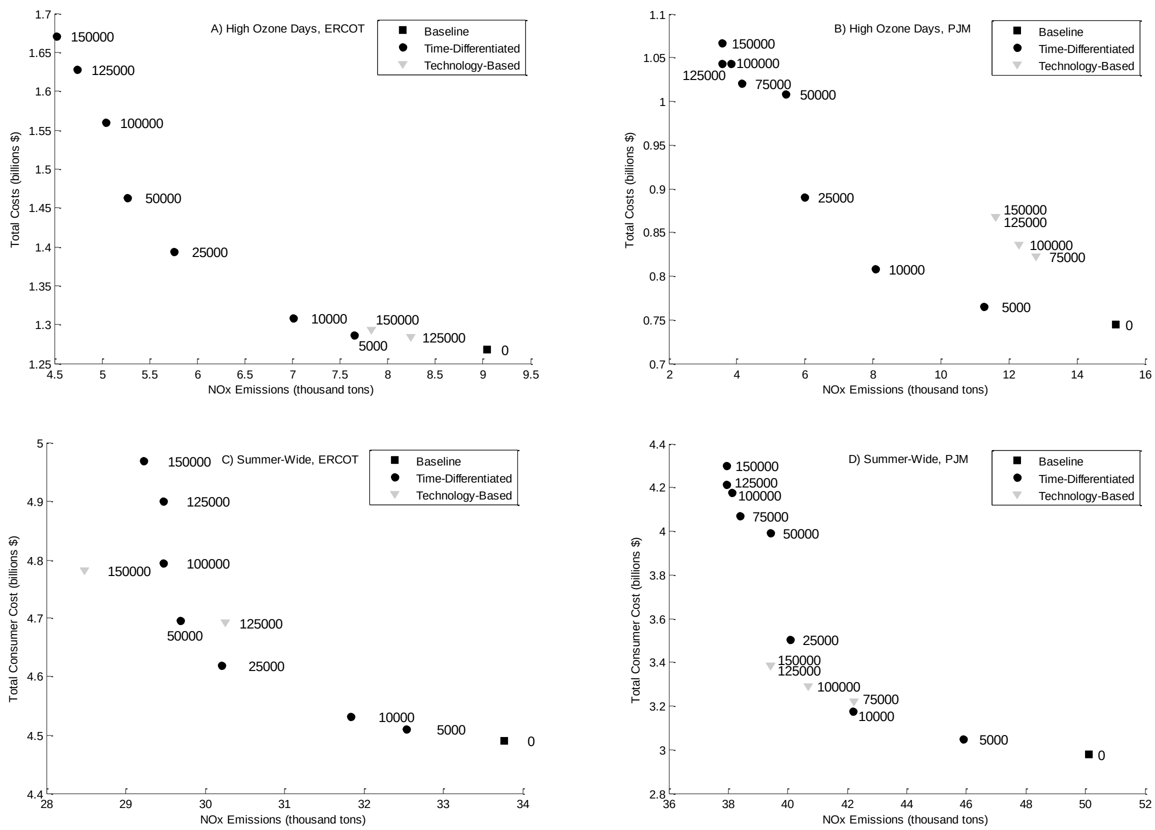
Figure 32: Percent change in electricity generation from baseline over the entire summer by coal-fired (black) and gas-fired (gray) generators in ERCOT (A) and PJM (B) under various time-differentiated prices (labeled with 'T' and the price in thousands \$/ton) and technology-based standards (labeled with 'F' and the time-differentiated price on which SCR installations are based).



8.2.3: Costs to Consumers

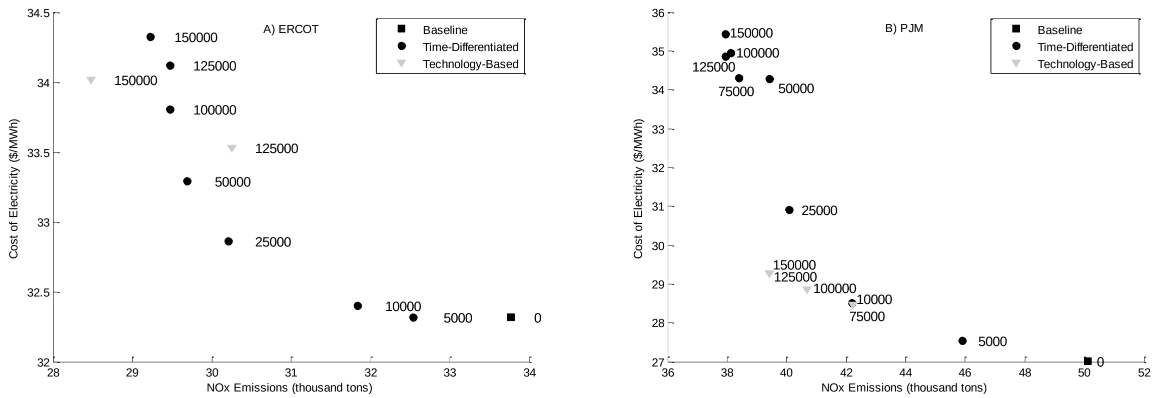
Considering costs to consumers instead of to producers highlights a different set of tradeoffs between time-differentiated pricing and technology-based standards in PJM, but not ERCOT. In ERCOT, time-differentiated pricing is again more cost-effective with respect to consumer costs for reducing summer-wide NOx emissions on high ozone days (Figure 33, A). It is also more cost-effective for low NOx emission reduction targets summer-wide, but not for high reduction targets (Figure 33, C). In PJM, as in ERCOT, time-differentiated pricing is the more cost-effective tool for summer-wide NOx emissions reductions (Figure 33, C). But whereas technology-based standards in PJM are less cost-effective at reducing summer-wide NOx emissions when considering producer costs (Figure 31, B), they are actually more cost-effective when considering consumer costs at the more stringent standards (Figure 33, D).

Figure 33: Costs to consumers versus NOx emissions in ERCOT (left) and PJM (right) on high ozone days (top) and over the entire summer (bottom). Time-differentiated prices are plotted as blue circles, whereas technology-based standards are shown by red triangles and labeled with the time-differentiated price on which the installed capacity of SCR was based.



Technology-based standards impose fewer costs on consumers at high summer-wide NOx reductions because they increase electricity (marginal) costs over the summer much less than time-differentiated pricing (Figure 34). Technology-based standards affect operations only at targeted coal plants, so may lead to redispatching of a limited number of plants. Time-differentiated pricing, though, affects operations at most or all plants on the system by imposing an additional cost on them. Redispatching under high time-differentiated prices is greater than under comparable technology-based standards, resulting in higher marginal costs and greater consumer costs under time-differentiated prices. But at low time-differentiated prices, redispatching occurs less than under technology-based standards, so costs rise less.

Figure 34: Average summer-wide electricity prices in ERCOT (A) and PJM (B) under time-differentiated prices (circles, labeled in $/ton) and technology-based standards (gray diamonds, labeled with the time-differentiated price on which the installed capacity of SCR is based).



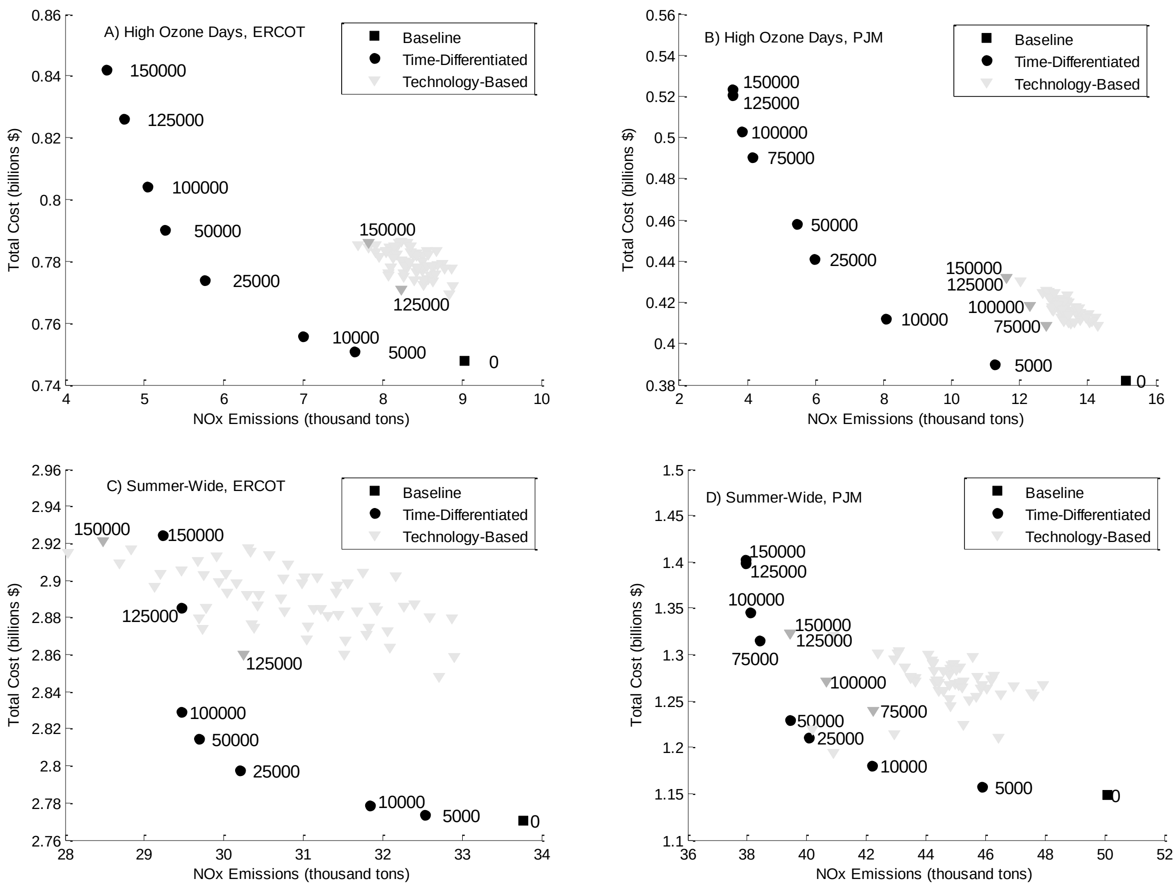
8.2.4: Randomly-Assigned Technology-Based Standards

In the results above, SCR installations are assumed to be required for only those plants that choose to install them under a comparable time-differentiated price. However, this is not the only possible allocation of SCR installation that could occur under a technology-based standard. Indeed, the rulemaking process is a complex one involving many stakeholders and competing interests (Keohane et al., 1998; McCubbins et al., 1987), making it difficult to predict which plants will be required to install SCR under a technology-based standard. This section presents emissions and costs under technology-based standards where SCR adoption mandates are allocated randomly to plants. Specifically, about 4.5 and 7 GW of operating SCR capacity are randomly allocated in ERCOT and PJM, respectively, equivalent to the installed capacity of SCR at $150,000 per ton time-differentiated prices in each system.

In ERCOT and PJM, emissions and costs on high ozone days do not vary greatly across the different SCR allocations (Figure 35, A and B). The technology-based standards where SCR is randomly allocated are clustered tightly. All of these points achieve less cost-effective emission reductions than a time-differentiated price, as previously observed.

There is much greater divergence in summer-wide emissions and costs among technology-based standards depending on how the SCRs are allocated. The majority of the technology-based standards are less cost-effective than time-differentiated pricing (Figure 35, C and D). In fact, none of the random allocations of SCRs in PJM achieve more cost-effective summer-wide NOx emission reductions than time-differentiated pricing (Figure 35, D). In ERCOT, though, there are a few possible allocations under which summer-wide NOx emission reductions are more cost-effective than under time-differentiated pricing (Figure 35, C). One of those allocations is the "optimal" allocation of SCRs in ERCOT, wherein SCRs are assigned to those plants that choose to install SCR under a time-differentiated price. The other allocations, though, are the results of random allocations of SCR that simulate the unpredictable rulemaking and implementation process. These allocations show that it's possible that a technology-based standard could achieve more cost-effective summer-wide emission reductions in ERCOT, but this result is highly uncertain. Moreover, it's highly unlikely, given that most allocations result in less cost-effective reductions than time-differentiated pricing. Thus, the choice between a time-differentiated price and technology-based standard in part turns on a key tradeoff between the *potential* and *certainty* of the cost-effectiveness of summer-wide emission reductions under each policy.

Figure 35: NOx emissions versus costs on high ozone days (top) and the entire summer (bottom) in ERCOT (left) and PJM (right) under various time-differentiated prices (circles, labeled in $/ton) and technology-based standards. Light gray triangles indicate technology-based standards in which SCR is randomly allocated to plants; roughly 4.5 and 7 GW are allocated in ERCOT and PJM, respectively, corresponding to the installed capacity of SCR in each system under a time-differentiated price of $150,000 per ton. Labeled dark gray triangles are also technology-based standards, but for these points, SCR is installed at those plants that chose to install them under the time-differentiated price corresponding to the point's label.



8.2.5: Summary and Discussion

Time-differentiated pricing offers a more cost-effective means of achieving any desired level of NOx emission reductions on high ozone days than a technology-based standard. This is also true in large part for summer-wide NOx emissions. Although technology-based standards at the highest installed capacities of SCR tested here can yield more cost-effective summer-wide emission reductions, this is only true for a small number of total possible SCR allocations. By considering the less predictable regulatory and appeals process, which could lead to suboptimal allocation of SCR, a time-differentiated price appears superior to a technology-based standard for reducing NOx emissions not just on high ozone days, but also over the entire summer. The one advantage observed here of technology-based standards is that they can impose lower costs on consumers. But costs are only slightly lower, and may not justify other losses that result from using technology-based standards.

8.3: Summary and Discussion of Tradeoffs between Considered Policies

The results presented in this chapter yield a number of useful insights into the tradeoffs between two commonly considered strategies for reducing NOx emissions, undifferentiated pricing (to simulate cap-and-trade) and technology-based standards, as compared to a third novel approach, time-differentiated pricing. Based on the criteria considered here –cost increases to producers and consumers and NOx emission reductions – time-differentiated pricing is the most cost-effective approach for reducing NOx emissions on high ozone days. Time-differentiated pricing can achieve a given level of NOx emission reductions on high ozone days at lower cost, both to producers and consumers, relative to either undifferentiated pricing or technology-based standards. This statement is true for costs just on high ozone days as well as over the entire summer. This result is therefore robust to the assumed variability in the proportion of summer days that are high ozone days. Time-differentiated pricing can also yield very large reductions in NOx emissions on high ozone days.

If the goal is to reduce the aggregate level of NOx emissions over a summer, however, undifferentiated prices offer the most cost-effective approach. Time-differentiated pricing as envisioned here not only has less cost-effective reductions, but also has a limited ability to reduce summer-wide NOx emissions because it is only active for a small fraction of summer days. A time-differentiated price is preferred to technology-based standards for moderate levels of summer-wide emission reductions, considering the lack of predictability in the allocation process. For much greater summer-wide emission reductions, technology-based standard may be more cost-effective, but would still be much less so than an undifferentiated price. Thus, based on the criteria considered here, there is little rationale for implementation of a technology-based standard in lieu of a market-based pricing scheme, either time-differentiated or undifferentiated. This finding is consistent with a wealth of studies that have demonstrated that market-based systems, such as an undifferentiated price, are a superior policy instrument than technology-based standards for other purposes (Carlson et al., 2000; Keohane, 2003; Krupnick et al., 2000; Schmalensee & Stavins, 2012; Tietenberg, 2010).

Jointly, these findings demonstrate that time-differentiated pricing could be an effective regulatory instrument for reducing ozone exceedances and increasing the number of regions that are in attainment with the ozone NAAQS. Moreover, they demonstrate that time-differentiated pricing may be more effective than instruments currently used towards those ends, namely technology-based standards and, in the eastern part of the U.S., cap-and-trade programs. The link between NOx emissions and ozone exceedances is explored further in the next chapter.

Chapter 9: Conclusion

In order to comply with legal requirements under the Clean Air Act and to mitigate public health impacts, peak ozone concentrations must be reduced across the U.S. This research assesses the cost-effectiveness of three types of policies at reducing NOx emissions that contribute to these high ozone concentrations: time-differentiated pricing, undifferentiated pricing and technology-based standards.

Time-differentiated pricing is found to be the most cost-effective option for reducing NOx emissions on high ozone days. Moreover, it can achieve substantial emission reductions of 50% or more on those days. These findings were consistent across the two power systems study here, which cover a large portion of the variability in power systems across the U.S.

In the absence of air quality modeling, it is not possible to draw any conclusions regarding reductions in high ozone concentrations, which is the indicator variable of ultimate concern. Nonetheless, assuming that NOx emissions on high ozone days contribute substantially to ozone concentrations on those days, then time-differentiated pricing is shown here to be the most cost-effective instrument for abating ozone on high ozone days. Some evidence for this assumption exists in (Sun et al., 2012), where NOx emission reductions on a high ozone day were found to reduce ozone concentrations on that same day.

However, over the entire summer, time-differentiated pricing offers little room for NOx emission reductions, and those reductions it does achieve are much less cost-effective than those achieved under an undifferentiated price. Thus, time-differentiated pricing is not a substitute for, but rather a complement to, existing regulations on NOx emissions. The implementation of time-differentiated pricing in this manner is discussed in the next section.

These findings contribute substantially to the body of knowledge on the merits of time-differentiated pricing as envisioned here. Prior studies established that time-differentiated pricing could be an effective tool for reducing NOx emissions on high ozone days (Bharvirkar et al., 2004; Sun et al., 2012). This work builds on this finding in many respects. First, it compares it to alternate policies for the first time, showing that time-differentiated pricing is not only an effective tool for reducing NOx emissions on high ozone days, but a more cost-effective one as well compared to two currently-implemented policies. This research also uses a more complex power system model for determining power generation and emissions and power plants, lending much more credibility to the extrapolation of these results to real-world power systems. In addition to using a more complex power system model, it also uses a novel approach to capture the short- and long-term effect of time-differentiating pricing on power plant operations, which is shown above to be crucial to fully understanding the effect of the policy on system-wide emissions and costs. (Sun et al., 2012) did not capture the long-term effect of inducing control technology installations, whereas (Bharvirkar et al., 2004) captured the long- but not short-term effect in a detailed manner.

9.1: Implementation of Time-Differentiated Pricing

Among the policies evaluated here, time-differentiated pricing was shown to be the most cost-effective method for reducing NOx emissions on high ozone days. However, its ability to reduce summer-wide NOx emissions was limited, and undifferentiated pricing achieved such reductions much cheaper. Thus, as recommended above, time-differentiated pricing should be layered on top of an undifferentiated pricing scheme.

This section discusses how a time-differentiated pricing scheme would be implemented in this way. First, the legality of time-differentiated pricing under the Clean Air Act is proved. Then time-differentiated pricing is formulated as a differentiated cap-and-trade program, which could take advantage of the existing cap-and-trade infrastructure. Finally, readily-available ozone forecasting models are discussed that would be essential to implementation of time-differentiated pricing in actual electricity markets.

9.1.1: Legality under the Clean Air Act

Time-differentiated pricing could be legally implemented by states as part of their requirement under the Clean Air Act to achieve compliance with the National Ambient Air Quality Standards (NAAQS). NOx, ozone and PM are all criteria pollutants under the Clean Air Act. As such, the EPA sets nationwide NAAQS for each pollutant with which states must comply. States that abide by the NAAQS are considered to be in attainment, whereas those that are not are considered to be in nonattainment. This determination is done on a pollutant-by-pollutant basis. Thus, a state may be in attainment for NOx and ozone, but in nonattainment for PM. For each pollutant NAAQS a state fails to attain, it must submit a State Implementation Plan ("SIP") to the EPA for approval. The SIP must detail how the state will implement, maintain and enforce the relevant NAAQS (42 U.S.C. § 7410(a)(1)). In plainer English, the SIP describes how the state will reduce and keep ambient concentrations of the relevant pollutant to levels below its NAAQS. Thus, under this section of the Clean Air Act, the EPA largely does not dictate pollution control measures to states. Rather, the Act allows states to choose the best method of emissions reductions.

To that end, states are provided significant flexibility under the Act in crafting a plan for reducing emissions. Acceptable control measures include "enforceable emission limitations and other control measures, means, or techniques (including economic incentives such as fees, marketable permits, and auctions of emissions rights)" (42 U.S.C. § 7410(a)(2)(A)). More specifically, states may implement "[m]arket-response strategies, which create one or more incentives for affected sources to reduce emissions, without directly specifying limits on emissions or emission-related parameters that individual sources or even all sources in the aggregate are required to meet" (40 C.F.R. § 51.493(a)(2)(ii)). A time-differentiated pricing scheme falls well within the bounds of this language. Furthermore, the EPA has historically given considerable leeway to states in devising SIPs (Monast et al., 2012), such as in approving the regional cap-and-trade approach adopted by states in the NOx SIP Call (Burtraw et al., 2014). Thus, historical precedents suggest the EPA would look favorably upon a market-based pollution reduction scheme, and moreover that a time-differentiated pricing scheme would satisfy regulations governing SIPs, allowing for its adoption by states.

There is one path by which the EPA may force a state to adopt a time-differentiated pricing scheme. Under the Clean Air Act, if a state fails to submit a SIP, submits a SIP that does not satisfy minimum criteria, or submits a SIP that the EPA disapproves, and fails to revise its SIP or submit a new one within two years, the EPA may promulgate a Federal Implementation Plan (FIP) for the state (42 U.S.C. § 7410(c)(1)). The same acceptable control measures that apply to SIPs apply to FIPs. Thus, the EPA could implement time-different pricing in states that fail to submit an adequate SIP.

9.1.2: Implementation of Time-Differentiated Pricing as a Differentiated Cap-and-Trade Program

Time-differentiated pricing could be implemented as a cap-and-trade program in one of two ways (Bharvirkar et al., 2004; Sun et al., 2012). One option is to differentiate the existing cap-and-trade program under the CAIR. Such a program is discussed at length in Section 3.4.1.1. Essentially, the trade-in ratio for each ton of emissions, i.e. the number of permits that would have to be surrendered per ton of emissions, would be increased on high ozone days from unity to whatever higher level is desired. In this manner, emissions on high ozone days would be priced much higher than others. Tweaking the existing program in this manner, though, would likely require higher allocations of annual permits, to account for the higher trading ratio, which could lead to unintended side effects.

Alternatively, an entirely new cap-and-trade program could be enacted that would apply just on high ozone days. At the beginning of each year, 'high-ozone' emission permits would be allocated to sources specifically for use on high-ozone days. The program's budget would be determined using the same model as the EPA uses for its other cap-and-trade programs, the Integrated Planning Model (U.S. Environmental Protection Agency Office of Air and Radiation, 2010). In addition to considerations that are normally made, such as for determining "ozone season" budgets in the current system, the expected number of high ozone days in a year would need to be forecasted with long-range meteorology models. However many permits allocated under this new program could be subtracted from the existing program to ensure aggregate NOx emissions do not increase.

Implementing time-differentiated pricing in either manner would be a challenging endeavor, given the many uncertainties that drive high-ozone episodes, but flexibility mechanisms could be built in to account for unexpected situations, e.g. a permit shortfall resulting from more high-ozone days occurring than were forecasted. The cap-and-trade program could take advantage of existing infrastructure for similar cap-and-trade programs, such as the Clean Air Interstate Rule for NOx trading. That infrastructure includes a trading platform, compliance mechanisms, and emission monitors and other data collection devices.

9.1.3: Availability of Day-Ahead Ozone Forecasting Models

One key tool for implementation of a time-differentiated pricing regime is day-ahead ozone forecasting capabilities. Without ozone forecasting, time-differentiated pricing could not be triggered until ozone concentrations already reached very high levels, largely obviating the potential benefits of time-differentiated pricing.

In competitive electricity markets, how far in advance ozone needs to be forecasted to implement time-differentiated pricing depends on when the day-ahead market of the relevant power system opens. Day-ahead markets are operated by system operators (ISOs) to determine the next day's dispatching. In the day-ahead market, each power plant submits an energy bid that states how much power the plant can generate hourly at a given price. In the case of time-differentiated pricing, costs of emissions would need to be included in this bid. As such, generators would need to know whether the time-differentiated price would be in effect the next day in order to submit their bids in the day-ahead market. Market operators already provide key information to power plants when the day-ahead market opens; whether the time-differentiated price will be in effect could be included in this information (ERCOT, 2013b).

Day-ahead markets open and close at different hours across power systems. ERCOT and PJM, the two power systems studied here, open their day-ahead markets at 6am (ERCOT, 2013b) and 12am (PJM Interconnect, 2014c), respectively, the day before the operating day for which generators submit bids. Thus, in the case of ERCOT, ozone conditions would need to be forecasted roughly 36 hours ahead. In PJM, an even longer forecasting range of roughly 40 hours would be required. Both forecasting range requirements are within the capacity of widely-used ozone forecasting tools. For instance, the Texas Commission on Environmental Quality provides ozone forecasts 3 days in advance (Texas Commission on Environmental Quality, 2014b), and the National Oceanic and Atmospheric Administration provides a 40 hour ozone forecast (National Oceanic and Atmospheric Administration, 2014).

Another important concern is whether these tools can accurately forecast ozone concentrations. If forecasts miss a high ozone day, then time-differentiated pricing cannot abate ozone on that day. Conversely, if a high ozone day is erroneously forecasted, then needless costs may be incurred through the time-differentiated price. Prior research by (Sun et al., 2012) suggests that even taking into account errors in ozone forecasting tools, time-differentiated pricing can still be a cost-effective tool for reducing ozone relative to other potential abatement strategies. Furthermore, ozone forecasting tools are continually improved, so current forecast errors should shrink over time.

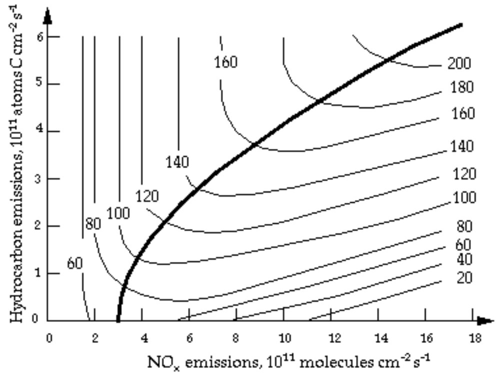
9.2: Implications for Air Quality

This research assesses the effect of policies on emissions and costs. It does not translate changes in emissions to changes in air quality, and ultimately to public health outcomes. As demonstrated below, this translation is a highly involved process that would have taken prohibitively-long to complete within the time allotted for the completion of this work. However, this research is part of a larger collaboration with researchers led by Dr. Elena McDonald-Buller at the University of Texas in Austin who will conduct air quality modeling based on emissions generated through this work. Instead of a full presentation of air quality impacts, this section provides a preliminary discussion of how emissions changes observed in this work may translate to air quality impacts.

NOx, as discussed in Section 2.2.1, is a requisite precursor to tropospheric ozone formation, and also contributes to particulate matter concentrations via reaction with various compounds. In regards to particulate matter, in general less NOx emissions mean less particulate matter formation. To what extent, though, is a highly complex process driven by many variables relating to atmospheric conditions and the presence of other compounds (U.S. Environmental Protection Agency, 2012d). Moreover, the proportion of particulate matter that stems from NOx varies greatly between regions (U.S. Environmental Protection Agency, 2012d). The main conclusion that can be drawn regarding particulate matter, therefore, is that policies that reduce NOx emissions more will likely lead to lower particulate matter concentrations.

Unfortunately, a similar relationship does not hold for NOx and ozone because ozone formation depends not just on NOx concentrations, but also VOC concentrations (Figure 36). In fact, in VOC-limited regimes (below the bold line in Figure 36), reducing NOx emissions can actually increase ozone concentrations. Most downtown urban areas are VOC-limited (U.S. Environmental Protection Agency, 2006). As a result, it is possible that greater NOx emission reductions achieved for the same cost under time-differentiated than undifferentiated pricing could actually lead to *increased* ozone concentrations and *negative* public health outcomes. Whether this occurs depends on numerous variables like meteorological conditions and the concentrations of various compounds, including VOCs.

Figure 36: Ozone concentrations (lines) as a function of NOx and hydrocarbon emissions. The thick line separates two ozone-forming regimes: above is NOx-limited and below is VOC-limited (Jacob, 1999b).



These uncertainties could be partially circumvented if one assumed that emission reductions would be achieved at the same point and time, just in different quantities, under two policies. But this is not true for time-differentiated and undifferentiated pricing. For one, the timing of emission reductions is very different. Undifferentiated pricing reduces NOx emissions over the entire summer, including on the days leading up to an ozone episode, i.e. a string of high ozone days. Conversely, time-differentiated pricing would only begin to reduce NOx emissions on the first day of an ozone episode. The difference in when reductions occur may not matter that much, since tropospheric ozone has a lifetime of one to two days at ground level and ozone concentrations are depleted overnight (Bloomer et al., 2010; Fowler et al., 2008). Indeed, (Sun et al., 2012) show that NOx emission reductions achieved on high ozone days reduce ozone concentrations on that same day. In light of these facts, same-day (or even day-before reductions in the case of a two-day ozone episode) reductions achieved under time-differentiated pricing may be just as effective as emission reductions in the days before an ozone episode.

Where emission reductions occur under these two policies will also be different to some extent. In general, both lead to substitution of gas- for coal-fired generation, so coal plants as a whole would be expected to generate less power. But which coal plants generate less will almost certainly be different under the two pricing regimes, e.g. depending on ramp constraints and minimum load requirements. Similarly, which gas plants make up for lower coal-fired generation will vary between the two policies. Consequently, it is not possible to say where ozone formation will be affected by emission reductions in each policy.

9.3: Limitations and Future Work

There are several limitations to this work and opportunities for future work. As discussed at length previously, the greatest opportunity for future improvement is linking changes in emissions to air quality and public health benefits. Carrying out this work would enable a holistic comparison of policies on the basis of costs *and* benefits, which is crucial when choosing the most efficient regulatory instrument.

The power system modeling used in this research could also be improved upon in several ways. First of all, control technology installation profitability is tested over a subset of summer weeks for just one year, but control technologies are a multi-year decision. This research does not capture long-term uncertainties or changes in power systems that could undermine the profitability of control technology installations. One such factor may be continued renewables expansion, which could depress electricity prices. Future work should address the long-term profitability of control technologies, such as through dynamic programming.

Also, when calculating emissions and costs, the unit commitment model is run for a full week in this research. However, unit dispatching in a real power system is done on a rolling day-ahead basis. Running the unit commitment model in this manner would provide a more realistic depiction of how redispatching would occur under a time-differentiated price. Outcomes may be different under this approach because the dispatching optimization would be done over a much shorter window of time within which generator constraints, like ramping limits and minimum down time, may interact differently.

Chapter 10: References

Acclaim Energy Advisors. (2014). ERCOT Sets New Winter Peak Demand- Real Time Prices Hit $5,000/MWh Cap on January 6, 2014. Retrieved March 01, 2014, from http://info.acclaimenergy.com/energyinsiderblog/bid/104410/ERCOT-Sets-New-Winter-Peak-Demand-Real-Time-Prices-Hit-5-000-MWh-Cap-on-January-6-2014

Ansari, A. S., & Pandis, S. N. (1998). Response of Inorganic PM to Precursor Concentrations. *Environmental Science & Technology*, *32*(18), 2706–2714. doi:10.1021/es971130j

Baldick, R., & Baughman, M. L. (2003). An Oligopoly Simulation of a Restructured ERCOT: Will Future Prices Be Competitive? *The Electricity Journal*, *16*(3), 59–71. doi:10.1016/S1040-6190(03)00029-0

Batlle, C. (2013). Electricity generation and wholesale markets. In I. Perez-Arriaga (Ed.), *Regulation of the Power Sector*. London: Springer.

Bell, M. L., Peng, R. D., & Dominici, F. (2006). The Exposure–Response Curve for Ozone and Risk of Mortality and the Adequacy of Current Ozone Regulations. *Environmental Health Perspectives*, *114*(4), 532–536. doi:10.1289/ehp.8816

Berman, J. D., Fann, N., Hollingsworth, J. W., Pinkerton, K. E., Rom, W. N., Szema, A. M., … Curriero, F. C. (2012). Health benefits from large-scale ozone reduction in the United States. *Environmental Health Perspectives*, *120*(10), 1404–10. doi:10.1289/ehp.1104851

Bharvirkar, R., Burtraw, D., & Krupnick, A. (2004). *NOx Emissions Trading and Episodic Air Quality Violations In Maryland*.

Bloomer, B. J., Vinnikov, K. Y., & Dickerson, R. R. (2010). Changes in seasonal and diurnal cycles of ozone and temperature in the eastern U.S. *Atmospheric Environment*, *44*(21-22), 2543–2551. doi:10.1016/j.atmosenv.2010.04.031

Burtraw, D., & Evans, D. A. (2003). *The Evolution of NOx Control Policy for Coal-Fired Power Plants in the United States*. Resources for the Future. Discussion Paper 03-23.

Burtraw, D., Linn, J., Palmer, K., & Paul, A. (2014). *The Costs and Consequences of Clean Air Act Regulation of CO2 from Power Plants*. Resources for the Future. Discussion Paper 14-01.

Carlson, C., Burtraw, D., Cropper, M., & Palmer, K. (2000). *Sulfur Dioxide Control by Electric Utilities: What Are the Gains from Trade?* Resources for the Future. Discussion Paper 99-44-REV.

Correia, A. W., Pope, C. A., Dockery, D. W., Wang, Y., Ezzati, M., & Dominici, F. (2013). The Effect of Air Pollution Control on Life Expectancy in the United States: An Analysis of 545

US Counties for the Period 2000 to 2007. *Epidemiology*, *24*(1), 23–31. doi:10.1097/EDE.0b013e3182770237.The

Dumas, J. (2013). Emerging Issues in North American Markets. *ERCOT*. Retrieved from http://apex.pjm.com/~/media/pjm-apex/presentations/20131030/20131030-panel-1-pm-dumas-ercot.ashx

Ellerman, A. D. (2003). *Are cap-and-trade programs more environmentally effective than conventional regulation?* MIT Center for Energy and Environmental Policy Research. Working Paper 03-015.

Ellerman, A. D., & Montero, J.P. (1998). The Declining Trend in Sulfur Dioxide Emissions: Implications for Allowance Prices. *Journal of Environmental Economics and Management*, *36*(1), 26–45. doi:10.1006/jeem.1998.1033

EME Homer City Generation, L.P., v. U.S. Environmental Protection Agency, et al., 21 Aug. 2012. U.S. Court of Appeals for the District of Columbia Circuit. Case No. 11-1302. (n.d.).

ERCOT. (2011). 2010 Demand and Energy Values by Weather and Congestion Zones. Retrieved March 01, 2014, from http://planning.ercot.com/reports/demand-energy/2010/

ERCOT. (2012a). *Capacity, Demand and Reserves Report, 2009 and 2012*.

ERCOT. (2012b). Demand and Energy - 2012: Interval Generation by Fuel Type. Retrieved April 01, 2014, from http://planning.ercot.com/reports/demand-energy/2012/

ERCOT. (2013a). 2013 Demand and Energy Values by Weather and Load Zones. Retrieved October 09, 2013, from http://planning.ercot.com/reports/demand-energy/

ERCOT. (2013b). ERCOT Nodal Protocols.

Fann, N., Fulcher, C. M., & Baker, K. (2013). The recent and future health burden of air pollution apportioned across U.S. sectors. *Environmental Science & Technology*, *47*(8), 3580–9. doi:10.1021/es304831q

Fann, N., Lamson, A. D., Anenberg, S. C., Wesson, K., Risley, D., & Hubbell, B. J. (2012). Estimating the national public health burden associated with exposure to ambient PM2.5 and ozone. *Risk Analysis : An Official Publication of the Society for Risk Analysis*, *32*(1), 81–95. doi:10.1111/j.1539-6924.2011.01630.x

Fowler, D., et al. (2008). *Ground-level ozone in the 21st century: future trends, impacts and policy implications*.

Fowlie, M., & Muller, N. (2013). *Market-based Emissions Regulation when Damages Vary across Sources: What Are the Gains from Differentiation?* National Bureau of Economic Research. Working Paper 18801.

GAMS Development Corporation. (2013). The GAMS Development Corporation Website. Retrieved from http://www.gams.com/

Hahn, R., & Stavins, R. (1991). Incentive-Based Environmental Regulation: A New Era from an Old Idea? *Ecology Law Quarterly*, *18*(1), 1–42.

Hubbell, B. J., Hallberg, A., McCubbin, D. R., & Post, E. (2005). Health-Related Benefits of Attaining the 8-Hr Ozone Standard. *Environmental Health Perspectives*, *113*(1), 73–82. doi:10.1289/ehp.7186

Jacob, D. J. (1999a). Oxidizing Power of the Troposphere. In *Introduction to Atmospheric Chemistry*. Princeton University Press.

Jacob, D. J. (1999b). Ozone Air Pollution. In *Introduction to Atmospheric Chemistry*. Princeton University Press.

Jaramillo, P., Griffin, W. M., & Matthews, H. S. (2007). Comparative life-cycle air emissions of coal, domestic natural gas, LNG, and SNG for electricity generation. *Environmental Science & Technology*, *41*(17), 6290–6. Retrieved from http://www.ncbi.nlm.nih.gov/pubmed/17937317

Keohane, N. O. (2003). *What Did The Market Buy ? Cost Savings Under the U . S . Tradable Permits Program for Sulfur Dioxide*.

Keohane, N., Revesz, R., & Stavins, R. (1998). The Choice of Regulatory Instruments in Environmental Policy. *Harvard Environmental Law Review*, *22*, 313–367.

Krupnick, A., Mcconnell, V., Cannon, M., Stoessell, T., & Batz, M. (2000). *Cost-Effective NOx Control in the Eastern United States*. Resources for the Future. Discussion Paper 00-18.

Levy, J. I., Baxter, L. K., & Schwartz, J. (2009). Uncertainty and variability in health-related damages from coal-fired power plants in the United States. *Risk Analysis : An Official Publication of the Society for Risk Analysis*, *29*(7), 1000–14. doi:10.1111/j.1539-6924.2009.01227.x

Martin, K. C. (2008). *Implementing a Time- and Location-Differentiated Cap-and-Trade Program: Flexible Nitrogen Oxide Abatement from Power Plants in the Eastern United States*. Massachusetts Institute of Technology.

Maryland Department of the Environment. (2012). Historical Air Quality Data: 8-Hour Ozone Exceedance Days. Retrieved April 23, 2014, from http://www.mde.state.md.us/programs/Air/AirQualityMonitoring/Pages/HistoricalData.aspx

Mauzerall, D., Sultan, B., Kim, N., & Bradford, D. (2005). NO emissions from large point sources: variability in ozone production, resulting health damages and economic costs. *Atmospheric Environment*, *39*(16), 2851–2866. doi:10.1016/j.atmosenv.2004.12.041

McCubbins, M., Noll, R., & Barry R. Weingast. (1987). Administrative Procedures as Instruments of Political Control. *Journal of Law, Economics, and Organization*, *3*, 243–277.

Monast, J., Profeta, T., Pearson, B. R., & Doyle, J. (2012). Emissions From Existing Sources : Section 111 (d) and State Equivalency. *Environmental Law Reporter*, *42*, 10206–10215.

Monitoring Analytics LLC. (2013). *State of the Market Report for PJM, Volume 2: Detailed Analysis*.

Morales-españa, G., Member, S., Latorre, J. M., & Ramos, A. (2013). Tight and Compact MILP Formulation of Start-Up and Shut-Down Ramping in Unit Commitment. *IEEE Transactions on Power Systems*, *28*(2), 1288–1296.

Muller, B. N. Z., & Mendelsohn, R. (2009). Efficient Pollution Regulation: Getting the Prices Right. *American Economic Review*, *99*(5), 1714–1739.

Muller, N. Z. (2011). Linking Policy to Statistical Uncertainty in Air Pollution Damages. *The B.E. Journal of Economic Analysis & Policy*, *11*(1).

National Oceanic and Atmospheric Administration. (2014). Air Quality Forecast Guidance - CONUS Area. Retrieved April 02, 2014, from http://airquality.weather.gov/sectors/conus.php?period=4#tabs

Olson, M. (1965). *The Logic of Collective Action: Public Goods and the Theory of Groups*. Cambridge, MA: Harvard University Press.

Patiño-Echeverri, D., Morel, B., Apt, J., & Chen, C. (2007). Should a coal-fired power plant be replaced or retrofitted? *Environmental Science & Technology*, *41*(23), 7980–6. Retrieved from http://www.ncbi.nlm.nih.gov/pubmed/18186326

Paul, A., Burtraw, D., & Palmer, K. (2009). *Haiku Documentation: RFF's Electricity Market Model, Version 2.0*. Resources for the Future.

PJM. (2012). Operational Analysis: Capacity by Fuel Type 2012. Retrieved April 23, 2014, from http://pjm.com/~/media/markets-ops/ops-analysis/capacity-by-fuel-type-2012.ashx

PJM Interconnect. (2012). Operational Analysis: Wind Generation. Retrieved March 01, 2014, from http://www.pjm.com/markets-and-operations/ops-analysis.aspx

PJM Interconnect. (2014a). Historical Metered Load Data. Retrieved March 01, 2014, from http://www.pjm.com/markets-and-operations/ops-analysis/historical-load-data.aspx

PJM Interconnect. (2014b). Maps: Transmission Zones. Retrieved March 01, 2014, from http://www.pjm.com/documents/maps.aspx

PJM Interconnect. (2014c). *PJM Manual 11 : Energy & Ancillary Services Market Operations*.

PJM Resource Adequacy Planning Department. (2013). *PJM Load Forecast Report*. Retrieved from http://www.pjm.com/~/media/documents/reports/2013-load-forecast-report.ashx

Reuters. (2014). Texas Grid Barely Avoids Outages as Cold Hits Utilities. Retrieved March 01, 2014, from http://www.reuters.com/article/2014/01/07/utilities-ercot-cold-idUSL2N0KG10G20140107

Richards, G., Weiland, N., & Strakey, P. (2006). Combustion Strategies for Syngas and High-Hydrogen Fuel. In R. Dennis (Ed.), *The Gas Turbine Handbook* (pp. 203–208). National Energy Technology Laboratory.

Schmalensee, R., & Stavins, R. (2012). *The SO2 Allowance Trading System: The Ironic History of a Grand Policy Experiment* (No. 12-44). Cambridge, MA. doi:10.3386/w18306

Seinfeld, J. H. (1989). Urban Air Pollution: State of the Science. *Science*, *243*, 745–752.

Sisternes, F. J. De, & Webster, M. D. (2013). *Optimal Selection of Sample Weeks for Approximating the Net Load in Generation Planning Problems* (No. 2013-03) (pp. 1–12).

State of North Carolina v. U.S. Environmental Protection Agency, 23 Dec. 2008. U.S. Court of Appeals for the District of Columbia Circuit. Case No. 05-1244. (n.d.).

Stavins, R. (1998). What Can We Learn from the Grand Policy Experiment? Lessons from SO2 Allowance Trading. *The Journal of Economic Perspectives*, *12*, 69–88.

Stevenson, D. S., Dentener, F. J., Schultz, M. G., Ellingsen, K., van Noije, T. P. C., Wild, O., … Szopa, S. (2006). Multimodel ensemble simulations of present-day and near-future tropospheric ozone. *Journal of Geophysical Research*, *111*(D8), D08301. doi:10.1029/2005JD006338

Sun, L., Webster, M., Mcgaughey, G., Mcdonald-buller, E. C., Thompson, T., Prinn, R., … Allen, D. T. (2012). Flexible NOx Abatement from Power Plants in the Eastern United States. *Environmental Science & Technology*, *46*(10), 5607–5615.

Swift, B. (2001). How Environmental Laws Work: An Analysis of the Utility Sector's Response to Regulation of Nitrogen Oxides and Sulfur Dioxide Under the Clean Air Act. *Tulane Environmental Law Journal*, *14*, 309–424.

Texas Commission on Environmental Quality. (2013). High Ozone in Your Metro Area. Retrieved April 23, 2014, from http://www.tceq.state.tx.us/cgi-bin/compliance/monops/ozone_summary.pl

Texas Commission on Environmental Quality. (2014a). Compliance with Eight-Hour Ozone Standard. Retrieved March 01, 2014, from http://www.tceq.texas.gov/cgi-bin/compliance/monops/8hr_attainment.pl

Texas Commission on Environmental Quality. (2014b). Today's Texas Air Quality Forecast. Retrieved April 02, 2014, from http://www.tceq.texas.gov/airquality/monops/forecast_today.html

Tietenberg, T. (1995). Tradeable Permits for Pollution Control when Emission Location Matters : What have We Learned ? *Environmental and Resource Economics*, *5*, 95–113.

Tietenberg, T. (2010). Cap-and-Trade : The Evolution of an Economic Idea. *Agricultural and Resource Economics Review*, *39*(3), 359–367.

Tietenberg, T. H. (1990). Economic Instruments for Environmental Regulation. *Oxford Review of Economic Policy*, *6*(1), 17–33. doi:10.1093/oxrep/6.1.17

Tong, D. Q., & Muller, N. Z. (2006). Policy Analysis Integrated Assessment of the Spatial Variability of Ozone Impacts from Emissions of Nitrogen Oxides, *40*(5), 1395–1400.

U.S. Bureau of Labor and Statistics. (2013). Consumer Price Index, All Urban Consumers. Retrieved October 09, 2013, from ftp://ftp.bls.gov/pub/special.requests/cpi/cpiai.txt

U.S. Energy Information Administration. (2012). Form EIA-923 Detailed Data. Retrieved April 01, 2014, from http://www.eia.gov/electricity/data/eia923/

U.S. Energy Information Administration. (2013a). Form EIA-860 Detailed Data. Retrieved March 01, 2014, from http://www.eia.gov/electricity/data/eia860/

U.S. Energy Information Administration. (2013b). *Monthly Energy Review: September 2013*.

U.S. Energy Information Administration. (2014). *Electric Power Monthly with Data for January 2014, Table 1.1*.

U.S. Environmental Protection Agency. (2005). *Rule to Reduce Interstate Transport of Fine Particulate Matter and Ozone (Clean Air Interstate Rule); Revisions to Acid Rain Program; Revisions to the NOx SIP Call, 70 Federal Register 91 (12 May 2005), pp. 25162-25405.*

U.S. Environmental Protection Agency. (2006). *Air Quality Criteria for Ozone and Related Photochemical Oxidants* (Vol. I).

U.S. Environmental Protection Agency. (2010). *Integrated Science Assessment for Particulate Matter*. Research Triangle Park, NC.

U.S. Environmental Protection Agency. (2011a). *2010 Progress Report: Emission, Compliance and Market Analyses*.

U.S. Environmental Protection Agency. (2011b). *Federal Implementation Plans: Interstate Transport of Fine Particulate Matter and Ozone and Correction of SIP Approvals, 76 Federal Register 152 (8 Aug 2011), pp. 48207-48712.*

U.S. Environmental Protection Agency. (2011c). *Progress Report 2011: Clean Air Interstate Rule, Acid Rain Program, and Former NOx Budget Trading Program*. Retrieved from http://www.epa.gov/airmarkets/progress/ARPCAIR11_downloads/ARPCAIR11_analyses.pdf

U.S. Environmental Protection Agency. (2011d). *The National Emissions Inventory*. Retrieved from http://www.epa.gov/ttn/chief/net/2011inventory.html

U.S. Environmental Protection Agency. (2012a). Air Markets Program Data: Query: Acid Rain Program (ARP), Continuous Emissions Monitoring Data. Retrieved October 09, 2013, from http://ampd.epa.gov/ampd/

U.S. Environmental Protection Agency. (2012b). Air Markets Program Data: Query: Acid Rain Program, Facility Attributes. Retrieved April 01, 2014, from http://ampd.epa.gov/ampd/

U.S. Environmental Protection Agency. (2012c). eGRID: eGRID 2012 Version 1.0. Retrieved October 09, 2013, from http://www.epa.gov/cleanenergy/energy-resources/egrid/index.html

U.S. Environmental Protection Agency. (2012d). *Regulatory Impact Analysis for the Final Revisions to the National Ambient Air Quality Standards for Particulate Matter*. Research Triangle Park, NC.

U.S. Environmental Protection Agency. (2013a). Green Book: 8-Hour Ozone Nonattainment Areas (2008 Standard). Retrieved March 31, 2014, from http://www.epa.gov/airquality/greenbook/map8hr_2008.html

U.S. Environmental Protection Agency. (2013b). NOx RACT Summary. Retrieved March 01, 2014, from http://www.epa.gov/region1/airquality/noxract.html

U.S. Environmental Protection Agency. (2013c). The Green Book Nonattainment Areas for Criteria Pollutants. Retrieved March 31, 2014, from http://www.epa.gov/airquality/greenbook/

U.S. Environmental Protection Agency. (2014). Air Markets Program Data, Acid Rain Program. Retrieved March 31, 2014, from http://ampd.epa.gov/ampd/

U.S. Environmental Protection Agency Office of Air and Radiation. (2010). *Documentation for EPA Base Case v.4 .10 Using the Integrated Planning Model*.

U.S. Environmental Protection Agency Office of Air Quality. (1999). *Guideline for Developing an Ozone Forecasting Program*.

Viscusi, W. K., Vernon, J., & Harrington, J. (2005). *Economics of Regulation and Antritrust* (5th ed.). Cambridge, MA: The MIT Press.

World Health Organization. (2000). *Air Quality Guidelines - Second Edition*. *Air Quality Guidelines - Second Edition*.

Zhang, R., Tie, X., & Bond, D. W. (2003). Impacts of anthropogenic and natural NOx sources over the U.S. on tropospheric chemistry. *Proceedings of the National Academy of Sciences*, *100*(4), 1505–9. doi:10.1073/pnas.252763799

Appendix A

A.1 Unit Commitment Generator Constraint Parameter Values

The following table contains the values for generator constraints included in the unit commitment model for both ERCOT and PJM. For prime movers, CC stands for combined cycle, GT for gas turbine, HY for hydro turbine, IC for internal combustion engine, OT for other, ST for steam turbine, WT for wind turbine, and SL for photovoltaic panel.

Prime Mover	Fuel Type	Min Down Time (hr)	Min Up Time (hr)	Ramp Rate (%/hr)	Start-up Fixed Cost ($/MW)	Start-up Fuel Use (MMBtu/MW)	Min Output (% of capacity)
CC	Gas	8	12	1	12.5	2.5	50
GT	Biomass	1	1	1	33	0	95
GT	Gas	1	1	1	12.5	0	0
HY	Hydro	1	1	1	0	0	0
IC	Oil	1	1	1	33	0	95
IC	Biomass	1	1	1	33	0	95
IC	Gas	1	1	1	12.5	0	95
OT	Gas	1	1	1	12.5	0	50
OT	Other	1	1	1	33	16.7	95
OT	Oil	1	1	0.3	33	16.7	50
ST	Biomass	12	16	1	33	16.7	50
ST	Biomass	12	16	1	33	16.7	50
ST	Coal	12	16	0.6	33	16.7	25
ST	Gas	12	16	1	33	16.7	30
ST	Nuclear	24	24	0.1	74	7.4	95
ST	Other	12	16	1	33	16.7	25
ST	Other	12	16	1	33	16.7	25
ST	Oil	12	16	0.3	33	16.7	25
ST	Other	12	16	1	33	16.7	25
ST	Coal	12	16	0.6	33	16.7	25
ST	Biomass	12	16	1	33	16.7	25
WT	Wind	1	1	1	0	0	0
SL	Solar	1	1	1	0	0	0

A.2 Calculating Dispatch Capacity of Combined Heat and Power (CHP) Plants

The capacity of combined heat and power (CHP) units was determined with Form 923 from the Energy Information Administration (U.S. Energy Information Administration, 2012). Form 923 lists the amount of power sold to the grid (“retail sales” and “resale sales”) by each CHP unit annually. Data from Form 923 for 2010, 2011 and 2012 were used to capture inter-year variability in operations. CHP units that sold less than 30% of their power to the grid in every year were considered industrial units and excluded from our analysis. Units that sold above 70% in any year were considered fully dispatchable. The remaining units, which sold less than 70% in every year but sold more than 30% in at least one year, were considered semi-dispatchable; their maximum capacity was reduced in half and they were dispatched in our model.

A.3 Parameters for SCR Installation

The below tables present the parameters used to simulate SCR installation. In addition to these parameters, SCR installation is modeled as reducing a generator's NOx emissions rate by either 90% or to a minimum threshold of 0.06 lb/MWh. Data is obtained from (U.S. Environmental Protection Agency Office of Air and Radiation, 2010).

Heat Rate of Coal-Fired Generator (MMBtu/MWh)	Heat Rate Penalty (%)	Variable O&M ($/MWh)
9,000	0.54	1.15
10,000	0.56	1.24
11,000	0.59	1.33

	Capacity of Coal-Fired Generator (MW)									
	100		300		500		700		1000	
Heat Rate of Coal-Fired Generator (MMBtu/MWh)	Capital Cost (1,000 $/MW)	Fixed O&M (1,000 $/MW-yr)	Capital Cost (1,000 $/MW)	Fixed O&M (1,000 $/MW-yr)	Capital Cost (1,000 $/MW)	Fixed O&M (1,000 $/MW-yr)	Capital Cost (1,000 $/MW)	Fixed O&M (1,000 $/MW-yr)	Capital Cost (1,000 $/MW)	Fixed O&M (1,000 $/MW-yr)
9,000	221	2.5	177	0.8	163	0.7	155	0.5	147	0.4
10,000	240	2.5	193	0.8	178	0.7	169	0.5	162	0.4
11,000	258	2.5	209	0.8	193	0.7	184	0.5	176	0.4