ISSUE BRIEF

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Introduction

The US Environmental Protection Agency (EPA) is preparing to finalize new air quality standards for ground-level ozone, which is commonly known as smog. Considerable controversy has surrounded the potential costs of these standards, with most circling around how EPA and its detractors value the "unknown" mitigation measures that will be needed to meet tighter standards. We argue that although the costs are, by definition, highly uncertain, they are likely to be closer to EPA's estimate than some interest groups and studies supported by such groups have claimed.

Ground-level ozone forms when volatile organic compounds (VOCs) react with nitrogen oxides (NO_x) in the presence of sunlight. Ozone can cause respiratory problems and lead to morbidity (such as asthma attacks) and premature mortality, among other negative effects. Under the Clean Air Act, EPA sets primary National Ambient Air Quality Standards for pollutants harmful to public health and states are responsible for developing plans to meet these standards. The current primary ground-level ozone standard is set at 75 parts per billion (ppb). EPA last set the standard in 2008, and the Clean Air Act requires EPA to review it every five years. EPA missed its five-year deadline in 2013 and is under court order to complete its review by October 1, 2015.

High ozone levels affect a large swath of the US population. About 123 million people, or 40 percent of the US population, currently live in areas with levels that exceed 75 ppb (2010 population; EPA 2015). EPA and its advisory committee of health experts say that health effects occur at even lower

Key Points

- The US Environmental Protection Agency is preparing to finalize a primary ground-level ozone standard below the current standard of 75 parts per billion (ppb).
- Although costs are not considered formally when setting the standards, the political implications very much depend on them. Costs will be considered formally when implementing the standards.
- Setting standards below 75 ppb may raise production costs at oil and gas wells, but critics largely overstate the costs.
- Although we take issue with some of the cost estimation assumptions made by EPA and other analysts, EPA's cost estimates are likely to be closer to the mark than those made by opponents of a tighter standard.

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levels and that the standard should therefore be tightened to protect human health within "an adequate margin of safety" (42 US Code § 7409(b)). Even more people live in areas that exceed these lower ozone levels (EPA 2014).[2]

But further tightening the standards below 75 ppb would impose additional costs to the US economy and, as noted, the controversy has largely focused on this. The Clean Air Act prohibits EPA from considering costs when setting standards, but when proposing and finalizing any costly regulations EPA is required by Executive Orders to report cost estimates in a regulatory impact analysis (RIA). In 2014, EPA proposed a new standard in the range of 65 to 70 ppb and estimated that the national costs (excluding California) of reaching 65 ppb in 2025 relative to the existing 75 ppb standard would be $15 billion (2011$), with benefits 1.3 to 2.5 times greater than costs (EPA 2014).[3]

In stark contrast, in a report for the National Association of Manufacturers, Harrison et al. (2015b) of NERA Economic Consulting estimate that the direct costs of achieving a 65 ppb standard would amount to between $75 billion and $85 billion in 2025.[4,5] Their study makes different assumptions than EPA about control costs and the quantity of emissions reductions needed. A recent study by Fisher et al. (2015) of Synapse Energy Economics, which was prepared for Earthjustice, criticizes a number of assumptions in Harrison et al. (2015b). The Fisher et al. cost estimates are similar to those of EPA.

In addition to debate surrounding the estimation of unknown cost, controversy extends to the effect of the proposed standard on oil and natural gas production. EPA modeling suggests that some rural locations, where gas and oil production raises ozone levels during winter months, may newly exceed 65 ppb. Some have suggested that a tighter standard therefore could harm domestic energy production, and given the growing importance of domestic energy production (CEA 2015), these effects could spill into the broader economy.

This paper focuses on the costs, rather than the benefits, of achieving the 65 ppb standard because the costs have been so hotly contested. We conclude the following:

- **Although the Clean Air Act prohibits consideration of costs when setting the standards, costs will be considered when implementing the standards and will continue to play into the public debate**. EPA and the states can consider costs when developing implementation plans. The long history of the Clean Air Act and academic literature provide no evidence that EPA has forced states to adopt policies with huge negative

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2 Most studies show that ozone affects health down to or near "background" levels (i.e., ozone concentrations in the absence of US emissions). As seen on TV commercials sponsored by the National Association of Manufactures (NAM), such levels do include concentrations from other countries, such as China. Although in a sense China is contributing to the US ozone problem, it could also be a costless part of the solution if the government follows through on its pledges to reduce urban air pollution.

3 Throughout this brief we report dollar amounts in 2011 dollars, which is consistent with EPA's estimate of costs and benefits.

4 For comparability with EPA's RIA, this estimate includes only the engineering costs of abatement. The EPA costs exclude the costs of meeting 65 ppb in California, and Harrison et al. (2015b) include costs in California. This distinction explains only a small share of the overall difference between the two estimates.

5 Readers may have seen reference to over a trillion dollars of costs on NAM-sponsored TV commercials. These cost estimates are cumulative costs over the entire program period, not costs in any given year.

economic effects and that, if anything, costs (as well as benefits) tend to be overestimated in RIAs.

- **Setting standards below 75 ppb may raise production costs at oil and gas wells, but Harrison et al. (2014) largely overstate the costs**. When analyzing 60 ppb (which is tighter than EPA is currently considering), Harrison et al. (2014) assume in a sensitivity analysis that production will remain at 2020 levels. This overstates costs because a) abatement technologies exist for oil and gas wells, b) low-cost oil and gas resources are available in regions expected to meet the standards, and c) oil and gas extraction does not occur exclusively in areas projected to exceed the standards.

- **Although we take issue with some of EPA's cost estimation assumptions, we find that the agency's cost estimates are likely far closer to the mark than those made by Harrison et al. (2015b).** Harrison et al. (2015b) inappropriately exclude the Clean Power Plan from their baseline, raising estimated costs by roughly $4.6 billion per year.[6] They also arbitrarily exclude some known controls identified by EPA. In addition, evidence from the recent literature and modeling of the power sector suggest that the costs of reducing NO_x emissions are likely much lower than assumed by Harrison et al. (2015b). Using our preferred assumptions to estimate the costs of implementing EPA's unknown controls reduces the Harrison et al. (2015b) estimate by more than 50 percent. Our estimates are fairly close to those of EPA and Fisher et al. (2015).

Overview of the Proposed Standards and EPA Cost Estimates

In 2014, EPA proposed a ground-level ozone standard between 65 and 70 ppb. Standards of 65 and 70 ppb differ greatly in geographic scope, cost, and benefit. While 70 ppb will result in higher benefits and costs in areas already affected by the current 75 ppb standard, 65 ppb will generate benefits and costs in new areas.

As part of its analysis, EPA performs a complex modeling exercise to forecast which counties will violate current and proposed alternative ozone standards in the year 2025, and after 2025 in California. EPA projects that 13 counties will not meet 70 ppb in 2025 (Figure 1). These 13 counties are limited to four regions of the country that are already expected to exceed the current 75 ppb standard—setting the standard at 70 ppb would not cause any new areas to exceed the standard.

By contrast, an additional 67 counties would not meet the alternative proposed standard of 65 ppb. This standard would require new areas, including areas in the West (some of them rural), to develop implementation plans. In the areas projected to be furthest from meeting the standard, state or federal governments may choose to establish tighter controls on key sources of the precursors to ozone, VOCs and NO_x. The potential for these policies to span multiple sectors explains the heavy advertising and pointed statements arguing against a tighter ozone standard from the American Petroleum Institute and the National Association of Manufacturers (which paid for the studies by Harrison et al. [2014; 2015a; 2015b]).

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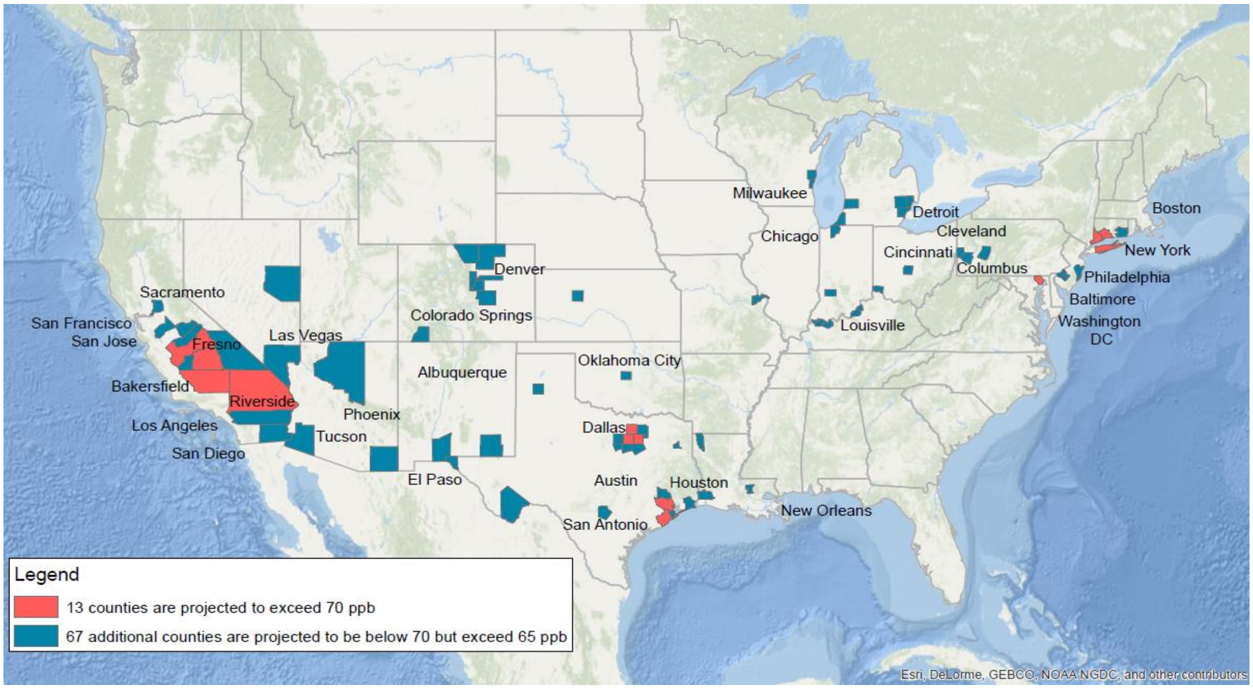
[6] The RIA for the proposed Clean Power Plan reports estimated NO_x reductions. This estimate does not quantify the portion of reductions that will occur in areas exceeding the standards, so the cost estimate we make here may be overstated.

Because meeting the tighter standard brings benefits to so many additional people, EPA projects that the net benefits of the 65 ppb standard will exceed the net benefits of the 70 ppb standard, even though costs are larger to meet the 65 ppb standard. Whereas 70 ppb results in net benefits of $2.5 billion to $9.1 billion, 65 ppb results in net benefits of $4 billion to $23 billion.

Figure 1. Areas Projected to Exceed 70 and 65 ppb[7]



Source: Data from EPA (2014).

To estimate the cost of the alternative proposed standards, EPA uses its air pollution model to estimate the NO_x and VOC emissions reductions needed to achieve them, starting from a baseline of 75 ppb. It then identifies and estimates the cost of a set of technologies that could be used to reduce emissions. For example, combustion controls such as low-NO_x burners reduce the formation of NO_x by lowering combustion temperatures (higher temperatures generally increase NO_x formation). Post-combustion controls, such as selective catalytic reduction, reduce NO_x emissions by removing it from exhaust gas.

Because these technologies do not reduce emissions sufficiently to meet the alternative standards nationwide, EPA then estimates the cost of the remaining "unknown" controls that must be implemented to meet the standards. The term "unknown" is somewhat misleading. Many controls and policies could be put in place to achieve these reductions; however, these options remain *unspecified* in EPA's analysis. In some areas, unknown (or unspecified) controls account for a significant portion of abatement; for example, they account for roughly 43 percent of NO_x emissions reductions needed to meet the 65 ppb standard in the East.

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[7] Compliance with the standards is projected for the year 2025 with the exception of California, which is projected for post-2025.

EPA assumes a constant abatement cost of $15,000 per ton for the remaining emissions reductions. This value is chosen to exceed the costs of nearly all applied known controls as well as nearly all NO_x credit prices that exist in areas that exceed existing standards. EPA justifies its estimate of a constant, rather than increasing, cost per ton by noting the challenges of estimating marginal abatement costs for future years based on current technology, especially due to the potential for policy-induced technological innovation and diffusion. In contrast, in the series of reports by Harrison et al. (2014; 2015a; 2015b), the authors argue, and we agree, that the assumption of constant abatement costs is not appropriate because abatement costs typically increase with additional abatement. Harrison et al. assume that marginal abatement costs increase linearly with abatement. They anchor their cost estimates with two hypothetical emissions reduction policies, as we discuss later in this paper.

The Role of Costs in Setting and Implementing Ozone Standards

Under the Clean Air Act, EPA is required to set ambient air quality standards that protect public health and the environment within an "adequate margin of safety." The courts have interpreted the Clean Air Act to imply that when establishing standards EPA is not permitted to consider costs of compliance.

After EPA sets the standards, states with areas violating it must create implementation plans that describe how they will reach the standards over a given period of time. States with areas further above the standards have more time to meet them than do states with areas closer to the standards, on one hand. On the other hand, those states with areas further above the standards have to adopt more stringent policies to meet the standards.

One way of thinking about the costs of meeting an ozone standard is to look backwards at what happened when EPA established or tightened standards. As Harrington (2006) notes, there are very few ex post analyses of EPA regulations generally, let alone of earlier ozone standards. Nevertheless, Harrington examines an Office of Management and Budget study that compares ex ante cost estimates appearing in 47 RIA's with ex post estimates of actual costs, looking deeply at a few case studies and at Harrington's own comparisons of 25 cases examined by Harrington, Morgenstern, and Nelson (2000). Unfortunately, none of these cases covers ozone, but most are for environmental regulations and are instructive. Harrington (2006) finds that RIAs tend to overestimate costs, but they also tend to overestimate benefits, with no bias in the ratio of benefits to costs. Applying these lessons to the proposed ozone standards, EPA's costs are likely to be overestimates, perhaps because they don't explicitly forecast technological change, but we might expect that the benefits could be lower than anticipated as well.

Implications for the Oil and Gas Industry

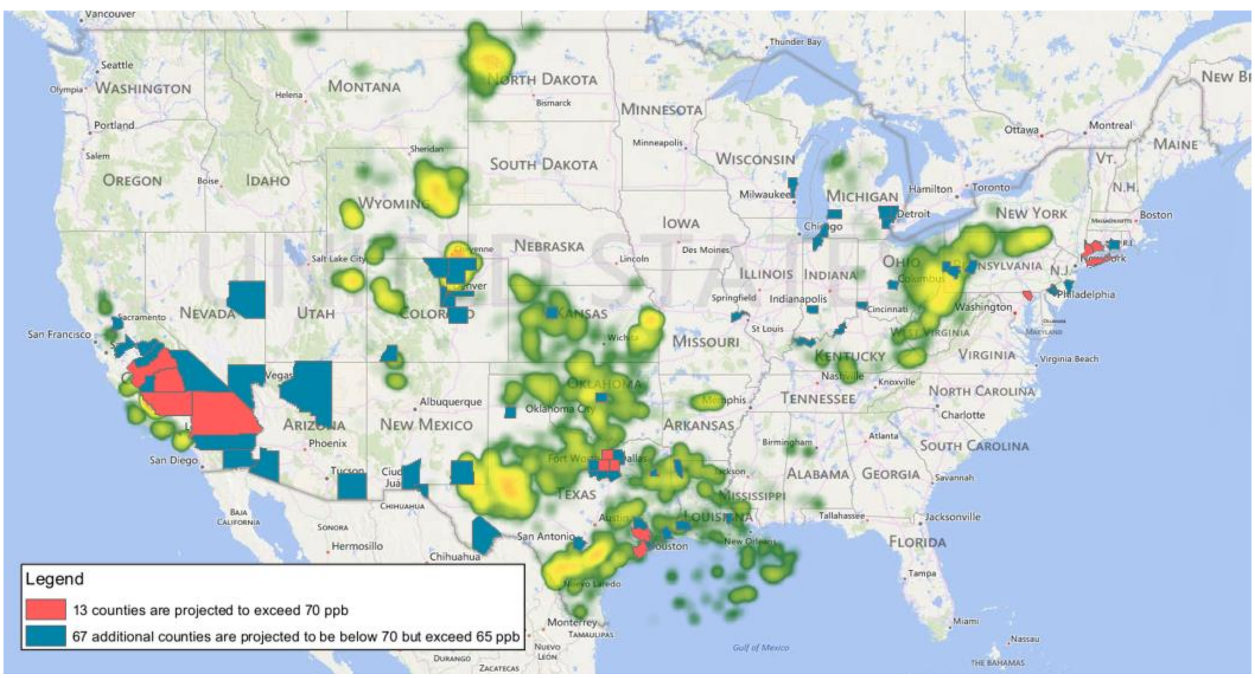
Setting standards below 75 ppb may raise production costs at oil and gas wells, but Harrison et al. (2014) likely largely overstates these costs. Under the alternative proposed standards, several areas where oil and gas production occurs will exceed 65 ppb. Due to the apparent, yet imperfect, correspondence between areas exceeding the standard and regions with oil and gas production (see Figure 2, below), concerns have been raised that the proposed standards will restrict oil and gas activity. While these concerns are understandable, we note several reasons why they may be overstated.

First, a sensitivity analysis in Harrison et al. (2014) assumes that 60 ppb would effectively end the growth of natural gas production, which would increase natural gas prices and harm consumer welfare (Harrison et al. 2015a and 2015b do not perform this analysis for 65 ppb.) Nothing in our reading of EPA's RIA for the new ozone standards or for the new methane and VOC proposed rules gives any indication that the costs of meeting the rules will be high enough to suspend production growth. In the case of the new ozone standards, other less-costly control measures, some of which are described in the next section, would be taken before suspending oil and gas production growth. In addition, EPA has finalized standards on new oil and gas methane sources as well as voluntary programs on existing sources, which will likely result in significant VOC reductions, with costs attributable to those programs rather than the ozone standards.

Second, oil and natural gas are produced in many areas where emissions do not cause the standards to be exceeded (Figure 2). Therefore, unless blanket emissions controls were instituted nationwide, these areas—where production is less damaging to the environment and public health—could increase production to offset decreases in or near areas exceeding the standards (but probably at some added costs).

Finally, the ozone standards must be considered in the context of international demand and supply conditions. Demand for oil and natural gas is highly volatile and difficult to project over long time horizons, and new supply sources may emerge in the coming years. Such shocks could cause a slowdown in domestic gas or oil production. Declining US oil and gas production under such circumstances would be costly to the US economy, but this illustrates the uncertainty faced by the industry, regardless of the ozone regulations.

Figure 2. Overlay of Counties Projected to Exceed 65 and 70 ppb Standards with Drilling Permit Activity in the Last 180 days



Source: Drilling permit activity heat map provided by Drillinginfo (http://info.drillinginfo.com/) and used with permission; accessed September 1, 2015.

Hypothetical Control Policies Imply Abatement Costs Similar to EPA's Estimates

Directly comparing the compliance cost estimates of EPA's RIA and Harrison et al. (2015b) is challenging because of significant differences in assumptions and methodology. In general, we question several important assumptions in Harrison et al. For example, Harrison et al. (2015b) omitted the Clean Power Plan from their analysis, wrongly attributing NO_x reductions and the associated costs to the ozone standards instead of the Clean Power Plan. Further challenging a direct comparison, Harrison et al. (2015b) and EPA model compliance costs in different years and in a few cases, the cost and timing assumptions presented by Harrison et al. are unclear or differ among their three reports.[8]

Because of these differences, it is not possible to fully disentangle the reasons why the unknown control cost estimates by Harrison et al. (2015b) exceed EPA's estimates by a factor of approximately five in 2025. Instead, we demonstrate the implications for estimated compliance costs by improving several of the key assumptions in the Harrison et al. analysis. First, we show that Harrison et al. (2015b) make assumptions that inappropriately increase the emissions reductions needed to meet 65 ppb. Second, we describe several feasible state or federal policies that could reduce emissions at much lower costs than Harrison et al. (2015b) assume. Finally, we compare compliance cost estimates using our preferred assumptions with cost estimates by EPA and Harrison et al. (2015b).

CORRECTLY DEFINING THE BASELINE

As noted, Harrison et al. (2015b) inappropriately exclude the Clean Power Plan from the baseline. Under the Clean Power Plan, states are required to meet greenhouse gas (GHG) emissions standards during the interim period of 2022 to 2029, and final standards that begin in 2030. States can reduce GHG emissions from the power sector by decreasing coal use and some combination of increasing gas-fired or renewables generation or adopting energy efficiency measures in power plants, businesses, or homes. These changes not only reduce GHG emissions but also reduce NO_x emissions.

EPA includes the proposed Clean Power Plan in the baseline against which emissions reductions needed to meet the ozone standards are measured. Because the agency includes the cost of reducing NO_x emissions in its cost analysis of the Clean Power Plan, counting costs to reduce those same emissions to meet ozone standards constitutes double-counting. Harrison et al. (2015b) argue that including the Clean Power Plan in the baseline is inappropriate because of uncertainty over how states will comply with it and because, traditionally, EPA only includes promulgated rules in the baseline. They therefore claim that EPA should include the costs of an additional 300,000[9] tons of NO_x emissions reductions (based on the proposed Clean Power Plan). This increases by 16 percent the emissions reduction needed to meet 65 ppb. However, these are not strong arguments for two reasons.

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[8] As noted by Fisher et al. (2015), EPA identifies approximately 200,000 tons of non-electricity generator emissions reductions from known controls, which are missing in Harrison et al. (2014; 2015a; 2015b). Where possible, we used cost assumptions from the most recent report. Following the authors' guidance, we refer to Smith et al. (2015) to infer some of the unspecified assumptions in Harrison et al. (2015b).

[9] Fisher et al. (2015) calculate a value of 309,000 from the RIA for the proposed Clean Power Plan. We use 300,000 here because we do not have a more precise number based on authors' guidance.

First, the costs and emissions reductions of any policy are uncertain, and it would be arbitrary to exclude the Clean Power Plan on that basis in this case only. Second, although EPA had not yet finalized the Clean Power Plan when it proposed the new ozone standards, the agency did so in August 2015. So EPA finalized the Clean Power Plan before finalizing the ozone standards, and it is therefore consistent with EPA practice to include the final Clean Power Plan in the baseline for the final ozone standards.

Estimating the cost implication of the decision of Harrison et al. (2015b) to omit the Clean Power Plan is tricky because including the Clean Power Plan may indirectly affect ozone compliance costs, for example, by affecting electricity prices. Nevertheless, to obtain a rough estimate of the effect of the omission on costs, we can use the average cost of power sector emissions reductions reported in Harrison et al. (2015a), of $15,000 per ton. A rough estimate of the effect of including the Clean Power Plan in the baseline indicates that costs would fall by $4.6 billion per year from those in Harrison et al. (2015b).

Several other important differences between Harrison et al. (2015b) and EPA (2014) are worth mentioning. First, Harrison et al. (2015b) perform their analysis using a 2022 instead of a 2025 baseline year, which raises their baseline emissions and total costs. Both of these baseline dates are reasonable. On one hand, areas that moderately exceed the standards would be required by EPA to come into compliance in 2022, although these deadlines are often delayed. On the other hand, EPA expects most controls to be in place in 2025, making 2025 representative of future compliance costs. Second, without explanation Harrison et al. (2015b) reclassified from known to unknown controls approximately 200,000 tons of NO_x emissions.

In aggregate, because of these assumptions Harrison et al. (2015b) assume 35 percent more emissions reductions are needed to achieve 65 ppb than does EPA (2014).

HYPOTHETICAL STATE AND FEDERAL POLICIES

Because EPA left open how such a large proportion of needed emissions reductions would be obtained (for example, roughly 43 percent of NO_x emissions reductions needed to meet the 65 ppb standards in the East), EPA gave outside interest groups opportunities to endorse their own approaches. Harrison et al. (2015b) specify two policies to achieve necessary emissions reductions: coal plant retirements and passenger vehicle retirements. While we appreciate their effort to specify and estimate the costs of these particular policies, we provide several examples of feasible policies for these sectors that reduce emissions at substantially lower cost.[10]

NO_x Cap and Trade in the Power Sector

To estimate the costs of achieving 65 ppb, EPA assumes that power plants can reduce NO_x emissions by employing selective catalytic reduction. The agency estimates that the average cost of these emissions reductions is $8,300 per ton.

However, there is a long history of using cap-and-trade programs to reduce NO_x emissions in the power sector, rather than specific technology mandates. Beginning in 1999, power plants and large industrial boilers have participated in a cap-and-trade program for NO_x emissions. The

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[10] Fisher et al. (2015) argue that energy efficiency investments could reduce NO_x emissions at low cost. However, to the extent that the Clean Power Plan places a binding constraint on power sector carbon dioxide emissions or emissions rates, energy efficiency is unlikely to substantially reduce carbon dioxide or NO_x emissions.

program initially covered the Northeast, expanded to the Southeast and Midwest in 2003, and expanded further in 2009 to cover about half of the country.

When implemented efficiently, cap-and-trade programs can reduce emissions at lower cost than installing selective catalytic reduction technology at all plants (as assumed by EPA) or retiring coal-fired plants (as assumed by Harrison et al. [2015b]). The cost advantage arises from the fact that the cap-and-trade program encourages emitters to find the lowest-cost emissions reduction opportunities. For example, the cost of installing selective catalytic reduction technology may vary across power plants, and only the lowest-cost plants would install it under the cap-and-trade program; other plants would find other ways to reduce emissions. Earlier and existing cap-and-trade programs could serve as a model for a future program to reduce NO_x emissions. Results from the RFF Haiku electricity model (Paul et al. 2009) suggest that a national cap-and-trade program could reduce emissions by 420,000 tons at an average cost of $7,100 per ton (for consistency with EPA's analysis, these estimates reflect simulations that include the Clean Power Plan but do not include mandatory selective catalytic reduction at all plants). This is in contrast to the Harrison et al. (2015b) estimate of $15,000 per ton.

Vehicle Retirement Program

To anchor their upper cost estimates, Harrison et al. (2015b) consider a hypothetical vehicle retirement program that offers individuals money to retire older vehicles, which would reduce emissions because older vehicles have higher NO_x emissions rates than newer vehicles.[11] For example, emissions are reduced when a vehicle meeting earlier emissions standards is retired and replaced by a vehicle meeting upcoming, tighter emissions standards. The marginal cost of reducing emissions is proportional to the value of the subsidy divided by the difference in emissions rates of the retired and new vehicles.

Assuming that the highest-emitting cars are retired first, and that the value of the subsidy is held constant, the marginal cost of reducing emissions outside the electric power sector increases with total emissions reductions. As described in Smith et al. (2015), Harrison et al. (2015b) estimate that the marginal cost of the first ton of emissions reductions is $29,000 per ton, which is based on their estimated cost from the power sector (i.e., assuming that vehicle retirements are the next lowest-cost option after coal plant retirements). The marginal cost of vehicle retirement rises to $52,000 per ton at 10 percent reduction of vehicle emissions, and a whopping $250,000 per ton at 40 percent reduction of vehicle emissions.[12]

These costs are much higher than the estimates in the recent literature on vehicle retirement programs. Sandler (2012) analyzes the vehicle buyback program in the California Bay Area, and estimates an average cost of reducing NO_x emissions of $31,000 per ton. Li et al. (2013) analyze the 2009 Cash for Clunkers program, which offered an average subsidy of $4,400 to retire a vehicle and replace it with a new one meeting certain fuel economy requirements. About 600,000 vehicles were retired under the program and Li et al. show that it increased the average fuel economy of new vehicles sold during the program. The results suggest an average cost for NO_x

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[11] The various reports use somewhat different assumptions but in all reports the level of the subsidy was chosen for consistency with the Cash for Clunkers program.

[12] We assumed Smith et al. reported vehicle retirement cost estimates in 2009$ similarly to the Harrison et al. (2014) report.

reductions of about $31,000 per ton (not accounting for the benefits of reducing emissions of greenhouse gases and other pollutants such as VOCs).

Three adjustments need to be made when adapting the estimated cost-effectiveness of a recent retirement program, such as Cash for Clunkers, to a hypothetical future vehicle retirement program. The first is to account for the fact that the on-road vehicle fleet will have lower NO_x emissions rates in the next decade than the fleet had during the Cash for Clunkers program. Because the per-ton cost of reducing emissions is inversely related to the emissions rate of the retired vehicle, allowing for this effect will raise the estimated cost of emissions reductions. Harrison et al. (2015b) correct for this effect.

The second is to account for the size of a case-study program relative to the hypothetical program. Cash for Clunkers retired about 600,000 vehicles, and Li et al. (2013) estimate that the program reduced NO_x emissions by about 90,000 tons. As discussed below, emissions reductions would have to be an order of magnitude greater to achieve 65 ppb. Presumably, scaling up the retirement program would raise costs.

Finally, the purpose of Cash for Clunkers was to provide economic stimulus during the recession and to improve the fuel economy of the on-road vehicle fleet. Reducing NO_x emissions was not an explicit objective of the policy, a fact that should be accounted for in estimating the cost-effectiveness of a hypothetical program. In principle, targeting a retirement program at NO_x emissions could reduce emissions at lower cost. For example, rather than providing retirement subsidies based on the fuel economy improvement between the new and retired vehicle as under Cash for Clunkers, the subsidy could be tied to the retired vehicle's NO_x emissions rate and recent miles traveled to more effectively target high-emitting vehicles.

Neither Harrison et al. (2015b) nor Fisher et al. (2015) adjust their cost estimates to address the latter two issues, although Fisher et al. (2015) note that targeting NO_x emissions would reduce costs. Instead, Harrison et al. (2015b) assume that the same $4,400 that was offered in Cash for Clunkers could be offered in a hypothetical future retirement program. This assumption has no empirical basis, however, and leads to an arbitrary cost estimate. For example, suppose the government had decided that $2,200 per vehicle provided enough of a subsidy in the Cash for Clunkers program to stimulate the economy. In the Harrison et al. (2015b) analysis, the cost of reducing emissions is directly proportional to the value of the subsidy. Consequently, if the government had chosen a lower subsidy level, their cost estimates would be one-half of the estimates used in their report. But we do not see why the government's decision about how much stimulus to provide should be relevant to the question of how much it costs to induce the retirement of vehicles with high NO_x emissions. We view the Harrison et al. (2015b) vehicle retirement cost assumption to be no less arbitrary than the EPA assumption about the cost of unknown controls.

Fuel Taxes

Because vehicle tailpipe standards set limits on grams of NO_x emissions per mile, states (or the federal government) could reduce emissions by introducing policies to reduce miles traveled. For example, raising the gasoline tax would increase the cost of driving and encourage people to drive less (and to purchase vehicles with higher fuel economy). A vast literature estimates the effects of fuel prices and taxes on miles traveled for light-duty vehicles, and a much smaller literature does the same for heavy-duty vehicles. For example, Li et al. (2014) estimate that an increase in a state's gas tax of $0.10 per gallon would reduce gasoline consumption by 1.7 percent. The effect of fuel costs on truck miles traveled is harder to estimate and there is a much smaller literature on which to draw, but based on our analysis of individual truck behavior, one might expect that an increase in the diesel tax of $0.10 per gallon would reduce truck miles traveled by 0.5 percent.

To quantify the effects of fuel tax increases, suppose all states increase their gasoline taxes by $0.10 per gallon. Accounting for changes in future NO_x emissions rates of the on-road vehicle fleet, the tax increase might reduce NO_x emissions in 2025 by 17,000 tons and VOC emissions by 12,000 tons. The tax increase would have other benefits, such as reducing distortionary taxes on labor and capital or avoiding the need for other tax increases (Carbone et al. 2013). These benefits would reduce the cost per ton of NO_x and VOC reductions; in fact, the cost could even be zero if the tax revenue is used efficiently.

California's Transportation Policies

California has recently implemented several programs (some of which are funded by revenues from cap-and-trade auction sales) that may reduce NO_x and VOC emissions. Although many of these programs were created with the goal of reducing GHG emissions, they also may serve as examples of policies that states adopt to reduce ozone levels.

Several of these programs focus on reducing emissions from vehicles. For example, California's Voluntary Accelerated Vehicle Retirement Program offers $950 to individuals who wish to retire vehicles that failed their last smog check (low-income consumers are paid $1,400). Although the program is also available for vehicles that passed recent smog checks, it is intended to target high-emitting vehicles (ARB 2015a). As another example, the Clean Vehicle Rebate Program encourages the purchase or lease of electric, hybrid, and fuel-cell vehicles by offering up to $4,800 in rebates per vehicle (ARB 2015b).

With the passage of California's Senate Bill (SB) 962, 60 percent of future auction revenue from the state's cap-and-trade program has been designated for transportation and sustainable communities programs. These funds support the development of a high-speed rail system, clean vehicle programs, and the expansion of public transit and affordable housing projects, among other goals. In addition, California recently passed a law that would reduce ozone-causing emissions by increasing the use of electric vehicles. By encouraging the use of alternative forms of transportation, these programs and others, such as SB 375 (which targets land use), are expected to reduce vehicle miles traveled and NO_x emissions.

USING OUR PREFERRED ASSUMPTIONS YIELDS COST ESTIMATES SIMILAR TO EPA

In this section, we modify the cost assumptions presented by Harrison et al. (2015b) to present an alternative cost estimate that still relies on their chosen methods of reducing emissions. The preceding discussion suggests that NO_x emissions could be reduced from the power sector at a cost of roughly $7,100 per ton. This estimate is about half that of the Harrison et al. (2015a) figure of $15,000 per ton. In addition, as discussed earlier, the cost of retiring light-duty vehicles is likely lower than assumed by Harrison et al. In contrast, EPA assumes a constant cost of $15,000 per ton for all emissions reductions by unknown controls.

To illustrate the implications of the differing cost assumptions, we estimate costs in a representative year (2025) because we (like EPA) do not attempt to model the dynamics of state implementation plans. We present the cost estimates that result from (1) EPA's calculations (2) Harrison et al.'s vehicle retirement program,[13] and (3) a power sector emissions reductions program and vehicle retirement program with our preferred cost assumptions.

Table 1. Estimated Cost of Using Unknown Controls to Reduce Power Sector and Vehicle NO_x Emissions by 750,000 tons

Method of estimation	Annual cost of unknown controls in 2025 (2011$)
EPA (2014)	$11 billion
Harrison et al.'s (2015b) vehicle retirement program	$38 billion
RFF calculations: Power sector trading program and modified vehicle retirement program	$12 billion

We begin with the EPA cost analysis. EPA assumes that NO_x emissions must be reduced by 750,000 tons using "unknown" controls. The average cost of these reductions is $15,000 per ton, yielding a total cost of $11 billion (first row in Table 1).

In the second row of Table 1, we present the cost estimate that results from using Harrison et al.'s (2015b) vehicle retirement program to reduce the same 750,000 tons. This is the cost estimate that Harrison et al. (2015b) would have produced if they had kept their own cost assumptions but had used EPA's assumptions about the emissions reductions needed to achieve 65 ppb. The estimate is $27 billion higher than EPA's—$38 billion for achieving 750,000 tons of emissions reductions using unknown controls.

Finally, in the third row of Table 1, we present the cost estimate calculated from reducing these 750,000 tons through the combination of a more realistic power sector emissions reduction program and a more realistic vehicle retirement program. To calculate this cost estimate, we replace Harrison et al.'s (2015b) cost assumptions with more realistic cost assumptions. Our construction of this cost estimate proceeds as follows.

First, we introduce power sector emissions reductions beyond those identified by EPA. Using the RFF Haiku model of a NO_x cap-and-trade program, we estimate that 220,000 tons more than EPA accounted for can be reduced from the power sector at an average cost of $7,100 per ton.[14] This relatively low cost estimate arises because market-based policies can reduce emissions at lower cost than command-and-control measures. Reducing these tons leaves a total of 530,000 tons (750,000–220,000) that must be reduced through another method.

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[13] We do not use the power sector emissions reductions presented by Harrison et al. (2015b) because they did not include the Clean Power Plan in their baseline.

[14] Strictly speaking, this $7,100 cost is the average cost for the entire NOx trading program, which reduces NOx by 420,000 tons.

We estimate the cost of reducing these remaining tons using our vehicle retirement marginal cost curve rather than Harrison et al.'s (2015b). Like Harrison et al., we use an increasing marginal cost curve; however, we modify the low and high endpoints of the curve to reflect our preferred cost assumptions. We anchor the curve at its lowest point at $7,100 per ton, which replaces Harrison et al.'s $29,000 to reflect the change from Harrison et al.'s coal retirement program to our trading program. We anchor the curve at its highest point at $94,000, which replaces Harrison et al.'s $250,000 for a vehicle scrappage program. We calculate this new maximum marginal cost by tripling the estimated costs of the Cash for Clunkers program by Li et al. (2013) to reflect the lower fleet emissions in 2025. Tripling the costs accounts for the reduction in average emissions rates of the fleet over time and assumes that the program causes the retirement of vehicles that have emissions rates roughly three times the fleet-wide average (which was the case under Cash for Clunkers). While this assumption is also arbitrary, it is at least anchored to observed cost-effectiveness. If the program were targeted to NO_x super-emitters, the cost of the vehicle retirement program would be even lower.

The last row of Table 1 shows our total cost estimate, $12 billion, which is the sum of the costs of reducing 220,000 tons from the power sector NO_x trading program (at $7,100 per ton) and reducing 530,000 tons from a more realistic vehicle retirement program at costs reflected in our new marginal cost curve. Our cost estimate is similar to the EPA estimate in the first row and much lower than the Harrison et al. (2015b) estimate in the second row.

Conclusions

Existing science suggests that reducing ozone levels will improve public health and the environment. EPA estimates that these benefits exceed the estimated costs of achieving the tighter ozone standards. In contrast, Harrison et al. (2015a; 2015b) conclude that EPA vastly understates costs because the unknown technologies and policies to obtain these reductions are likely to have much higher costs than EPA assumes. Here we show that using more appropriate assumptions yields estimates that are more consistent with EPA's estimates.

The analysis by Harrison et al. (2015b) and this paper are static in the sense that they do not account for technological progress. In its cost-benefit analysis, EPA notes that past regulations have preceded substantial and largely unanticipated technological progress, particularly when the regulations provided strong incentives for innovation. Such unanticipated innovation could cause actual costs to be lower than these estimates.

In addition, many cost-effective policy options outside the electric power sector could reduce costs, such as a vehicle retirement program that is targeted toward reducing NO_x emissions. Contrary to estimates by Harrison et al., but consistent with the pattern identified by Harrington (2006), we therefore find that—even though the method of estimating a fixed cost per ton for unknown controls is unsatisfying at best—the cost estimate presented by EPA is reasonable.

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