What Do Economists Have to Say About the Clean Air Act 50 Years After the Establishment of the Environmental Protection Agency?*

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Abstract

Air quality in the United States has improved dramatically over the past 50 years in large part due to the introduction of the Clean Air Act and the creation of the Environmental Protection Agency to enforce it. This essay is reflection on the 50-year anniversary of the Environmental Protection Agency, describing what economic research says about the ways in which the Clean Air Act has shaped our society - in terms of costs, benefits, and important distributional concerns. We conclude with a discussion of how recent changes to both policy and technology present new opportunities for researchers in this area.

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1 Introduction

Air quality in the United States has improved dramatically over the past 50 years. Since 1980, ambient levels of "criteria" air pollutants, those pollutants which are most consistently monitored by the Environmental Protection Agency (EPA) because they are known to be harmful for human health, have fallen by in many cases 85 or 90 percent, as shown in Figure 1. Some pollutants, such as lead in gasoline, were banned outright. This decline in pollution has occurred even while primary sources of air pollution like electricity generation, transportation, and industrial activity have continued to expand. How can air quality have improved so dramatically when the underlying sources of pollution have continued to grow? One important reason is the introduction of the Clean Air Act (CAA) and the creation of the EPA to enforce it, almost 50 years ago.

The 1970 Clean Air Act, combined with significant amendments in 1977 and 1990, and more recent changes that reflect the evolution of scientific consensus, have made the current reach of the CAA expansive. Some researchers and policy makers have described the CAA as one of the most significant federal interventions into markets in the postwar period (Greenstone 2002). How large are the costs that the CAA imposes on producers and consumers, both in absolute terms and per additional unit of pollution reduction? And how do the costs compare to the benefits? Economists and policy makers have attempted to provide answers to these questions since the CAA was first proposed.

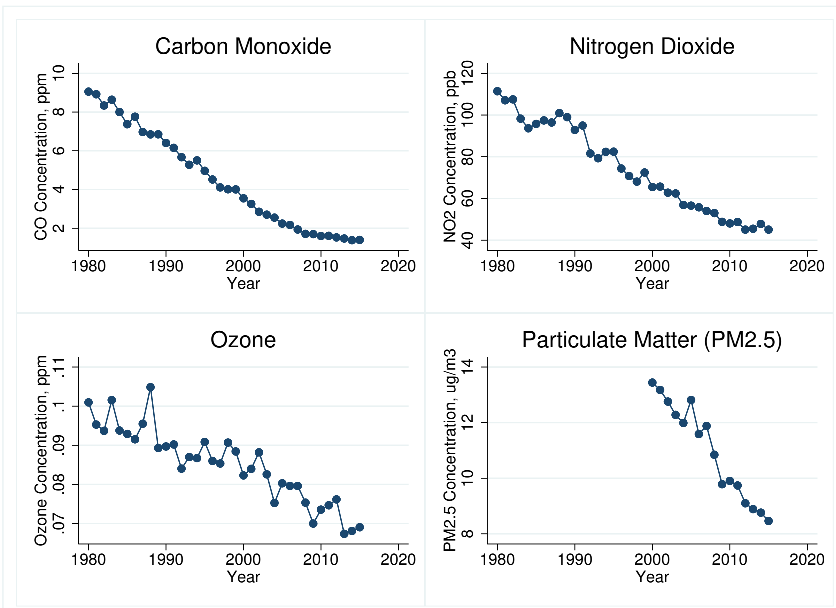
This essay is reflection on the 50-year anniversary of the EPA, describing what researchers and policy makers know about the ways in which the CAA has shaped our society - in terms of costs, benefits, and important distributional concerns. We begin with a short overview of how the legislation has evolved. We then discuss how the Clean Air Act has affected ambient air quality in the United States before turning to what the literature suggests about how to value these benefits. In the case of economic costs, we focus both on positive and normative findings - attempting to take stock of what we know about the effects of the Clean Air Act and also about how the Clean Air Act could be made more efficient or cost effective. We conclude with a discussion of how recent changes to both policy and technology present new opportunities for researchers in this area.

2 Background

The federal Clean Air Act was first implemented in 1963, but the original legislation provided limited federal oversight of state efforts and led to disappointing results. In response, Congress enacted the Clean Air Act Amendments of 1970 and established the Environmental Protection Agency, vastly increasing federal power to address the problem of air pollution.

The Clean Air Act of 1970 initially relied exclusively on "command and control" regulations that were set using criteria that focused on the health benefits of cleaner air without regard to the economic costs of cleanup. The law also gave sweeping powers to a new federal government agency, the Environmental Protection Agency, to mandate specific mitigation measures. The legislation focused on common, dangerous, air pollutants known as "criteria air pollutants." Today the criteria

Figure 1: Trends in Air Pollution in the United States, 1980 to 2015



NOTES: Source: Authors, using data from the Environmental Protection Agencys Air Quality System. Note: Each annual average is computed based on the criteria set under the National Ambient Air Quality Standards. Carbon monoxide is measured as the second-highest eight-hour average. Nitrogen dioxide is measured as the ninety-eighth percentile of daily maximum one-hour average. Ozone is measured as the fourth-highest daily maximum eight-hour concentration. PM2.5 is measured as an annual average; ppm is parts per million; ppb is parts per billion; ?g/m3 is micrograms per cubic meter. Source: Authors, using data from the Environmental Protection Agencys Air Quality System.

pollutants include sulfur dioxide, carbon monoxide, nitrogen dioxide, lead, particulates, and ozone. National ambient air quality standards set the maximum allowable ambient pollution levels, and these standards are enforced locally on an annual basis.

Each year in July, the Environmental Protection Agency determines the set of counties that are in "nonattainment" of a particular standard. State governments must develop a pollutant-specific "State Implementation Plan" describing how these nonattainment counties will be brought into compliance. If a state fails to act or develops an inadequate plan, the EPA can withhold federal funding for the state air pollution control program, highway construction, and the construction of sewage treatment plants. The EPA can also ban permits for construction of major new and/or modified sources of a pollutant. In addition, the EPA can impose its own federal plan on nonattainment counties if it deems the States plan as insufficient. The Environmental Protection Agency also sets industry-specific emissions standards for new or modified sources of pollution in nonattainment counties, whereby affected facilities are required to adopt "lowest achievable emissions rate" technologies. The EPA also regulates mobile sources including most motor vehicles under the Clean Air Act, wielding the power to impose testing and certication requirements for engines and to require specic fuel formulations and additives. Thus, the Clean Air Act gives the EPA sweeping powers. These powers are sufficiently broad that that even the threat of regulatory action has been associated with reductions in pollution (Keohane, Mansur, and Voynov 2009).

Since 1970, there have been two major amendments to the Clean Air Act; one in 1977, and one in 1990. These amendments largely maintained and extended the command-and-control approach to regulation, but the latter also introduced some more market-based approaches into the regulatory mix. The 1977 amendments introduced "New Source Review," a policy designed to regulate major new or modified sources of pollution in attainment counties, whereas facilities in those counties had not previously faced much regulatory scrutiny. In nonattainment areas, regulations were tightened such that any new stationary source of air pollution was required to offset their emissions by finding a willing counterparty in the same area to reduce their emissions one for one. These amendments also established major permit review requirements to ensure compliance with the air quality standards.

The 1990 amendments updated national ambient air quality standards and continued to broaden the Environmental Protection Agencys enforcement powers as well as created important new market-based mechanisms, including the sulfur dioxide allowance-trading program to address the threat of acid rain. In a far reaching development, the amendments mandated lead-free gasoline (as of 1995), established new auto gasoline reformulation requirements designed to produce cleaner-burning gasoline, set standards to control evaporative emissions from gasoline, and mandated that new gasoline formulations be sold from May to September in many states in an effort to reduce ozone. The amendments also required automobile manufacturers to further reduce tailpipe emissions from new vehicle fleets.

In another notable development extending EPA regulatory authority under the Clean Air Act, the 1990 amendments began regulating "toxic" air pollutants, identifying 189 hazardous air pollutants and requiring that the EPA establish emission standards that provide for "an ample margin of safety to protect public health" by minimizing the amount of toxic pollution they release into the air to the extent that technology allows.

Air quality rules have continued to evolve to reflect scientific consensus. For example, in 1997, standards were tightened for ozone, and particles less than 2.5 micrometers in diameter were regulated for the first time. The fine particle standard was revised downward in 2006 and 2012. Ozone regulations were further tightened in 2008 and 2015, respectively. In 2000, the EPA nalized a rule requiring additional reductions in automobile emissions along with requirements (called "Tier 2 standards") for cleaner automotive fuels. Beginning in the 2000s, the EPA also adopted provisions to prevent upwind areas from polluting downwind areas, particularly in the case of ozone. These regional policies started with the NOx Budget Trading Program (2003-2008), which evolved into the Clean Air Interstate Rule (2010-2015), which has subsequently evolved into the Cross-State Air Pollution Rule, governing ozone precursors and SO_2 emissions in the northeast United States.

To summarize, the Clean Air Act has given the federal Environmental Protection Agency sweeping powers to regulate virtually all industrial activities in the U.S. These powers have been extended over time through the 1977 and 1990 amendments as well as subsequent smaller regulatory changes. The existing regulatory framework is extraordinarily complex, potentially creating substantial regulatory burden for affected firms. These regulations have also led to substantial improvements in air quality and associated public health. We discuss each in more detail below.

In discussing this body of work, it is useful to remind ourselves what economic theory suggests to be key parameters that govern the efficiency or cost-effectiveness of environmental policy. The optimal level of pollution in society occurs where the marginal costs of reducing emissions by one unit are exactly equal to the marginal social benefit of that same reduction in air emissions. Thus, understanding and estimating marginal abatement costs and/or marginal social benefits would allow researchers and policy makers to have a better understanding as to which of the existing regulatory instruments are most cost effective and which we should reconsider. It would also allow us to have a better understanding as to whether existing regulations are too lax or too stringent. Economists are still far from being able to accomplish that goal for every regulatory component of the Clean Air Act, but it is useful to keep it in mind as we discuss existing studies that may be stepping stones on that path.

3 Benefits of the Clean Air Act: Falling Pollution, Improved Health

Estimates of the benefits of clean air legislation are built on two foundations: How much did the legislation reduce pollution? And how much does a reduction in pollution affect human health? Since the existing regulations affect certain areas more than others, economists have used this regulatory variation to construct a variety of counterfactuals for what air quality would be in the absence of the policy. For example, polluting firms in nonattainment counties are more heavily regulated than similar firms in other counties; attainment is based on whether a county is above or below a sharp cutoff for each pollutant; only some counties are in nonattainment in any given year; and counties move in and out of attainment status based on their pollution levels each year. Moreover, county attainment status is only partially in the control of county-level actors, because some air pollution can blow in from neighboring counties. This temporal and spatial variation facilitates air quality comparisons of affected and unaffected counties using research designs like regression discontinuity, difference-in-difference, and/or instrumental variables. Such research has provided compelling evidence as to the ways in which the Clean Air Act has shaped both air quality and population health over the past 50 years.

3.1 Effects of the Clean Air Act on Air Quality

How much of the dramatic reduction in air pollution over the past 50 years can be attributed to the Clean Air Act? Henderson (1996) was the first to recognize that the regulatory variation embedded into the Clean Air Act lends itself to exploring the causal effects of the policy. Henderson (1996) examined the effect of ozone regulations between 1977 and 1987, comparing changes in air quality in newly designated nonattainment counties to those that were in compliance with the ozone standard. He finds that a switch to nonattainment status reduced ozone concentrations by 3 to 8 percent, depending on the measure.

Considerable research has followed along similar lines. For example, Chay and Greenstone (2003a) explore the extent to which the original nonattainment designations in 1970 for total suspended particulates are associated with improved air quality in affected regions. They found a 10 percent improvement in air quality in the years after the regulations went into place in affected counties. Auffhammer et al. (2009) examine the impact of changes following the 1990 Clean Air Act Amendments, and they find that nonattainment designation led to air quality improvements (11-14 percent), but primarily in the set of communities directly adjacent to the violating air quality monitor. More recent work has explored the air quality improvements associated with the 1997 tightening of fine particulate standards, with Bishop et al. (2018) and Currie, Voorheis, and Walker (2019) finding statistically significant improvements in air quality of around 10 percent below baseline levels. In summary, there is a range of compelling evidence that nonattainment designations improve local air quality on average, and especially in areas where pollution is initially most severe.

Other research has focused on the regional air quality programs under the Clean Air Act, such as the NOx Budget Trading Program (Deschenes, Greenstone and Shapiro 2017), the SO_2 Acid Rain program (Barreca, Neidell and Sanders 2017; Chan et al. 2018), and the Clean Air Interstate Rule (Murphy 2017). The NOx Budget Trading Program operated a cap-and-trade system for over 2,500 electricity generating units and industrial boilers in the eastern and midwestern United States between 2003 and 2008. Using a difference-in-differences design, Deschenes et al. (2017) find a 40 percent decline in NOx emissions leading to a 6 percent reduction in ozone concentrations, with most of the benefits at the upper end of the pollution distribution (for example, the number of days with ozone above 65 parts-per-billion fell by 35 percent).

One potential issue with permit trading of pollution emissions is that the incremental damages from pollution may differ across locations, but most market-based regulations penalize emissions at the same tax rate or permit price regardless of location. These "undifferentiated" policies can have significant distributional consequences and have led to contentious debates. For example, Chan et al. (2018) argue that by allowing permits to be traded between relatively less populous areas with low abatement costs and relatively densely populated areas with high abatement costs, the SO_2 Acid Rain Trading program achieved lower gains in health than a counterfactual program that would have simply mandated equal reductions across units. In another example, Fowlie (2010) studies the introduction of the NOx budget program and the way that it interacted with electricity market deregulation and the restructuring of regional electricity markets occurring at that time. The harm from NOx is localized, so the distribution of pollution sources matters. Fowlie points out that under existing cost-plus utility regulation, electricity producers faced a known return on investing in costly pollution abatement equipment and would have stronger incentives to invest in abatement. In the set of electricity markets that were eventually deregulated, uncertain returns discouraged some low-abatement cost producers from investing in abatement, offsetting some of the possible efficiency gains of the emissions trading program. In the end, pollution abatement under this particular permit program was not concentrated in the highest marginal harm locations.

Looking forward, efficient policy design for pollutants with strongly local impacts requires a nuanced understanding of location-specific heterogeneity in damages as well as abatement costs. The considerable uncertainties associated with both damage and cost estimates also have direct implications for the optimal ex-ante design of these policies. For example, Fowlie and Muller (2013) explore whether market-based policies can be designed to deal with pollutants like NOx that are not "uniformly mixed." They show that when damages or abatement costs are uncertain, undifferentiated emissions trading policies can be better than differentiated policies that impose higher costs in more polluted areas.

Even though automobile emissions have historically accounted for a large fraction of criteria emissions, there has been comparably less research on the efficacy of various mobile emissions standards on overall air quality improvements. Overall, automobile emissions have clearly declined. For example, on-road vehicle emissions accounted for 70 percent of carbon monoxide emissions in 1970 and 31 percent in 2017. Similarly, on-road vehicle emissions accounted for 49 percent of volatile organic compound emissions (an ozone precursor) in 1970 and 11 percent in 2017, according to the National Emissions Inventory data (at https://www.epa.gov/air-emissions-inventories/air-pollutant-emissions-trends-data, accessed on January 15, 2019). However, relatively little research has sought to isolate the causal effect of mobile emissions standards on overall air quality improvements. The limited existing research generally describes model-year trends before and after standards change using a cross-section of vehicles (Kahn 1996), or describes data on abatement technologies (but not emissions levels) for different vehicle types (Bresnahan and Yao 1985). Part of the difficulty may be that emissions standards apply to the entire vehicle fleet in a given year, making it challenging for researchers to separate changes in tailpipe standards from other secular trends affecting ambient air quality.

The 1990 amendments to the Clean Air Act required the elimination of lead from gasoline. The available evidence suggests that the elimination of leaded gasoline reduced childrens blood lead levels dramatically in a short period of time (Aizer and Currie, 2019). The amendments also included provisions about reformulated gasoline that were intended to reduce ozone precursors (such as volatile organic compounds) from mobile sources. Reformulated gasoline is federally mandated in areas that are in severe nonattainment of the ozone standard. Less polluted areas which do not meet the ozone standard may opt into federal standards for reformulated gasoline as part of their plan to reach attainment. Auffhammer and Kellogg (2011) find limited evidence of air quality improvements in affected regions despite the fact that consumers must pay for the more expensive reformulated gasoline.

Finally, the CAA has promulgated a significant number of emissions standards for hazardous air pollutants through its air toxics regulatory program. As mentioned, these regulations require the EPA to develop maximum achievable control technology standards for over 180 toxic pollutants. While these standards are wide reaching, the large number of regulated pollutants has precluded comprehensive analysis by economists. Currie, Davis, Greenstone, and Walker (2015) use data from hazardous air pollution monitors to show the relationship between a subset of these regulated, toxic chemicals and distance from emitting facilities. They then use the openings and closing of these emitting facilities to better understand the ways in which these hazardous air pollutants affect local air quality and population health. They find significant effects of these plant openings/closings on measures of hazardous air quality, and ultimately population health, in the surrounding communities.

3.2 The Benefits of Pollution Reduction

Even if academics and policy makers perfectly understood how the CAA has affected air quality, it is unclear how we should value these improvements. A first step in understanding the value of air quality improvements is developing a better understanding of the various ways in which air pollution can affect societal outcomes. The next step, and one that is currently missing from the literature, is to translate these various impacts into a pollutant-specific damage function that can be used for policy analysis and valuation.

Economists have examined the effects of pollution on health, housing prices, and worker productivity, asking whether observed correlations are causal or whether they may be better explained by an omitted “third factor” such as low income. Researchers have developed a range of solutions to the problem of identifying causal effects, including using natural experiments and/or instrumental variables research designs to break the correlation between pollution exposure and omitted variables that might be correlated with both pollution and outcomes.

Work on the health effects of pollution has often focused on infants and young children, as well as the elderly. This focus is partly because these groups are thought to be particularly vulnerable.

In one of the first economic studies of this question, Chay and Greenstone (2003a) use the county-year variation in regulation, stemming from the initial implementation of the National Ambient Air Quality Standards, to compare the incidence of low birth weight and infant mortality in counties just above the nonattainment threshold and those just below when the law was introduced. They found that a regulation-induced, one unit decline in total suspended particulates led to 5 to 8 fewer infant deaths per 100,000 live births, relative to attainment counties. Almond, Currie, and Duque (2018) review the recent literature on the effects of air pollution on young children.

A number of studies have explored contemporaneous relationships between air pollution spikes and hospital admissions for respiratory disease. Ransom and Pope (1995) look at the effects of intermittent shutdowns of a large steel mill on children in the surrounding areas. Schlenker and Walker (2016) explore daily changes in pollution levels in areas around California airports. Daily spikes in air pollution do cause increases in admissions for asthma and respiratory illnesses, with large effects for both children and the elderly.

Deryugina et al. (2018) also explore the relationships between daily changes in pollution exposure and population health, exploiting changes in regional wind direction which drive daily pollution levels. They use claims data from Medicare, which covers all Americans over 65, and ask how daily spikes in PM2.5 affect mortality. An innovative feature of their study is that they look not only at the number of deaths, but they also estimate the number of life-years lost as a result of pollution. They do this by predicting individual life expectancy using each persons health history. They find that a reduction in fine particles of approximately 4 micrograms per cubic meter between 1999 and 2011 resulted in a gain of .011 per 1000, or a little over a month of life per elderly person. While these studies of pollution spikes and daily changes provide compelling evidence of harmful effects of pollution, they necessarily ignore some of the longer run implications of pollution on health outcomes.

A smaller but growing literature explores how increased pollution levels are likely to affect outcomes apart from direct health measures, including labor productivity, cognition (as measured by test scores), educational attainment, and even crime. See e.g. Graff-Ziven and Neidell (2013) for a comprehensive review of this literature.

Several studies examine the impact of short-term variations in pollution on labor supply and worker productivity. In an early paper on this topic, Graff-Zivin and Neidell (2012) examine piece-rate workers in California and find that variations in ozone have significant effects on both hours worked and productivity: An increase of ten parts per billion (on a mean of 48ppb) decreases worker hours by 20 minutes and reduces productivity by 0.12 standard deviations.

Isen et al. (2013) build on the Chay and Greenstone (2003a) study by following the cohorts who were born just before and just after the passage of the 1970 Clean Air Act into adulthood. Cohorts who were born before 1970, during levels of relatively high air pollution, look different on a range of outcomes, measured at age 30, compared to cohorts born after the relative improvements in air quality. They find that a decrease of 10 micrograms per cubic meter in total suspended particulates in the air breathed during pregnancy and early childhood was associated with a 1 percent increase in annual earnings at age 29 to 31 which works out to a lifetime income gain of \$4,300 (in 2008 dollars) per affected person. Most of this effect comes from increased labor force participation rather than changes in on-the-job earnings, suggesting that reductions in disability may be an important pathway.

Declines in airborne lead, due to the elimination of lead from gasoline, are also likely to have had far-reaching effects. Aizer et al. (2018) link preschool childrens blood lead levels to their future test scores and disciplinary records and find that declines in lead increased childrens test scores and reduced behavior problems and delinquency. Aizer and Currie (2019) focus specifically on lead from gasoline and show that individuals affected by the de-leading of gasoline were less likely to be delinquent or to commit crime, a finding that echoes earlier work using cohort-level analyses (e.g. Wolpaw-Reyes 2007).

3.3 Monetizing Benefits

The studies discussed so far establish that the Clean Air Act reduced pollution, and that pollution reductions have positive effects on health and well-being, as well as areas like worker productivity. But a benefit-cost analysis requires an additional step: putting a monetary value on the benefits. Attempts to put a dollar benefit on health improvements have typically involved two key ingredients: 1) estimates of the value of a "statistical life"; and 2) a concentration-response function relating pollution exposure and mortality risk.

The value of a statistical life is an estimate of how much people are willing to pay for small reductions in the risk of death. The EPA recommends that analysts use an estimate of \$7.4 million (in 2006 dollars) to quantify mortality risk reduction benefits, which is approximately the middle of the range of available estimates (for a review, see Viscusi and Aldy 2003). This "one-size-fits-all" estimate is potentially problematic given that pollution disproportionately affects the health of the very young and the very old, and willingness to pay for reductions in mortality risk may vary by age. Indeed, some researchers have begun using quality-adjusted life years (QALYs) in place of a single value of a statistical life measure (for example, Deschenes and Greenstone 2011; Deryugina et al. 2018).

Concentration-response functions describe how changes in air quality affect mortality risks or other measures of health and well-being. For example, the EPA uses two key studies to evaluate the benefits of proposed and existing PM2.5 reductions (Krewski et al. 2009; Lepeule et al. 2012). Researchers face a number of significant difficulties when estimating concentration-response functions for air pollution. First, the existing concentration-response estimates used by the Environmental Protection Agency are unlikely to represent causal relationships. They are based on cross-sectional data (e.g., cross-city comparisons) that control for some observable confounders, but there may be many other correlated, but unobserved, factors that impact both mortality and pollution. Second, significant biases may arise due to the difficulty of measuring an individuals air pollution exposure. Third, these studies primarily focus on mortality, ignoring other pernicious effects of air pollution. Fourth, pollutants are often highly correlated, and it is often difficult to empirically say which of the pollutants is responsible for the observed damages. Fifth, concentration-response functions potentially vary for different groups in the population. Sixth, the more compelling studies of concentration-response functions rely on high-frequency, short-run variations in pollution levelson the days when air pollution rises in a given city we can observe a corresponding change in mortality or morbidity rates. However, from a policy perspective, we are often most interested in how long-run changes in pollution map into population health and well-being.

Finally, if people take actions to avoid pollution exposure, then estimates of the effects of pollution that do not take such defensive, avoidance behavior into account will have a downward bias. For example, Moretti and Neidell (2011) show that taking account of the avoidance behavior generated by ozone alerts greatly increases the estimated effect of ozone emissions from the port of Los Angeles on emergency room visits and hospitalizations for respiratory problems. An additional consideration is that avoiding pollution is itself costly, and any reduction in avoidance costs should be counted as a benefit of pollution reduction.

For all of these reasons, monetizing the marginal benefits of pollution control remains challenging. Some researchers have tried to monetize health impacts using hospital costs for respiratory or cardiac admissions (e.g., Schlenker and Walker 2016). However, hospital reimbursement costs are poor measures of willingness to pay to avoid harm, which is ultimately what should enter into a proper concentration-response or damage function.

In a quest for a more encompassing welfare measure of air quality benefits, researchers have attempted to measure willingness to pay for air quality by using other methods such as those from stated or revealed preference studies. Stated preference methods, such as asking agents directly how much they value air quality (e.g. contingent valuation), have been criticized on a number of different grounds (Hausman 2012) except in cases of "passive use valuation" (Carson 2012). On the other hand, there has been an explosion of recent work on revealed preference measures to estimate willingness to pay for air quality. The most influential revealed preference approaches have come from hedonic housing value analyses. Researchers have shown how agents trading off housing prices with housing amenities (e.g. air quality) can provide information on willingness to pay for these amenities (Rosen 1974).

Starting with the groundbreaking work of Chay and Greenstone (2003), researchers have combined hedonic theory with causal methods to deliver a range of estimates of individual willingness to pay for air quality. For example, Chay and Greenstone (2005) use the national ambient air quality standards as an instrumental variable for air quality changes. Their instrumental variable estimates are much larger than ordinary least squares estimates and imply that each one unit reduction in TSPs increased home values by 0.7 to 1.5 percent. This translates into an estimated aggregate benefit of $4.5 billion per unit of total suspended particulate improvement for affected counties. There has been a subsequent explosion of work in the hedonic valuation literature, providing willingness to pay estimates for many different aspects of Clean Air Act regulatory improvements.

These revealed preference approaches all find that people value cleaner air and are willing to pay for it in terms of housing prices, though there is less consensus about how much. Moreover, there are legitimate concerns about these approaches. The assumption that environmental factors are capitalized into housing prices, while convenient, requires that people are fully aware of both the pollution levels and the effects of pollution. This is certainly truer in some contexts than others. If there are few housing sales in an area it may not be possible to track the underlying value of housing capital with any precision. People may not always be able to move in response to shifts in pollution, if for example, they are credit-constrained. Developing robust and generalizable approaches for recovering empirical measures of the welfare benefits associated with pollution control remains a high research priority.

A number of researchers have explored the monetary benefits of a single component of the Clean Air Act, but there have been few serious efforts to estimate the total benefits associated with the combined policies the EPA notwithstanding. For example, Chay and Greenstone (2005) suggest that regulation of total suspended particulates in the 1970s was associated with a $45 billion aggregate increase in housing values in nonattainment counties (relative to their attainment counterparts). Barreca, Neidell, and Sanders (2017) focus on the long-term effects of the SO_2 permit trading program introduced after the 1990 amendments to the Clean Air Act. Their design involves comparing adult mortality rates within a 100-mile radius of affected power plants to those in similar, unaffected counties. They estimate that the annual value of lives saved reached $134 billion per year by 2005, compared to program costs of approximately $3 billion per year.

The estimated benefits of cleaner air have been demonstrated to be large and significant. Yet key questions need further research: How much of the overall improvement in ambient pollution concentrations can be traced back to the Clean Air Act? In turn, how much of the reduction in infant mortality and of the overall increase in life expectancy can be attributed to the Clean Air Act rather than to improvements in medical care, living standards, and other factors? And have the benefits been uniform, or have some groups benefitted substantially more than others from clearing the air? Last, and perhaps most importantly, what does the damage function look like for each regulated pollutant, and what are the marginal social benefits of improving air quality even further?

4 Regulatory Costs of the Clean Air Act

From a social welfare perspective, the correct theoretical measure of the costs of environmental regulation is the (monetized) change in social welfare due to the reallocation of resources from the production of goods and services to pollution abatement activities (Hazilla and Kopp 1990). For this reason, private expenditures on compliance costs or engineering cost estimates are insufficient measures of economic costs, especially if there are significant general equilibrium impacts that extend beyond the directly regulated sector. Estimates of total costs should also include monitoring and enforcement.

Understanding the direct and indirect costs of the Clean Air Act is exceedingly difficult. It is hard to think of a credible counterfactual for what the US economy would look like if clean air legislation had never been enacted! Thus, the existing estimates of the economic costs of regulation have relied on a range of sources and methods that can broadly be classified into three categories: papers that seek to identify the ex-post, causal effect of the CAA on a range of different outcomes using methods similar to the papers on the effects of pollution and health described above; empirical industrial organization studies of a single industry that are used to estimate counterfactuals with and without regulation; computable general equilibrium models of the entire economy which are used to conduct counterfactual analyses of costs/output under different regulatory regimes. Each of these approaches has strengths and weaknesses which we will discuss below. An overall takeaway is that while we do not have a complete and accurate measurement of the total cost of the Clean Air Act, the current estimates suggest that the overall costs are likely to have been substantially less than the estimated benefits in terms of health and other outcomes.

4.1 Estimating Compliance Costs Based on Regulatory Variation in the Clean Air Act

Features of regulatory roll-out or design can be used to form counterfactuals for what regulated industries would have looked like in the absence of the program. For example, a number of different researchers have used the county-industry-year variation embedded in changes in air quality standards to estimate effects on a range of economic outcomes. Henderson (1995) shows that polluting industries in nonattainment counties exhibit lower growth rates, and that these effects are partly driven by the reallocation of industry to attainment counties. Kahn and Mansur (2010) similarly find that polluting firms tend to locate in areas not subject to the Clean Air Acts nonattainment designations. One countys loss may be another countys gain in terms of jobs and production, so that focusing only on short-run losses in nonattainment counties relative to attainment counties may be a misleading way to infer economic impacts, a point to which we will return.

Similarly, Becker and Henderson (2000) use plant-level data to examine the effects of ozone nonattainment status. From 1963 to 1992, they find a 25 to 45 percent drop in the number of new plant openings in nonattainment counties in polluting industries, relative to polluting industries in attainment counties. Focusing on two industries that are large emitters, they find that total plant operating costs are higher in ozone nonattainment counties. For example, plants in the industrial organic chemicals industry had 17 percent higher total operating costs in ozone nonattainment areas compared to similar plants in attainment areas. To summarize, there is wide-ranging evidence that these regulations have led to relative shifts in production away from nonattainment counties.

A series of papers have explored the extent to which these same regulations affect input demand for productive inputs like capital or labor. Greenstone (2002) is an early attempt to shed light on this question. He uses data from the 1967 to 1987 Census of Manufacturers to examine the extent to which Clean Air Act nonattainment designations affected plant input and output decisions. He estimates that in the first 15 years in which the Clean Air Act was in force (197287), nonattainment counties (relative to attainment ones) lost approximately 590,000 jobs, $37 billion in capital stock, and $75 billion (1987 dollars) of output in pollution-intensive industries.

From a social welfare perspective, the overall effect of "jobs lost," plant exit, or output losses in an area is unclear. If a worker loses her job due to a new regulation but finds a job tomorrow at the exact same wage, these costs may be minimal. In contrast, if the worker is unemployed for long periods of time and/or cannot find a comparable paying job in future years, these transitional costs of reallocating production may be quite large. There may also be capital adjustment costs and other allocative inefficiencies associated with reallocation. Walker (2013) investigates the transitional costs of the 1990 Clean Air Act Amendments for manufacturing workers by combining the county-pollutant-year regulatory variation of the changes in air quality standards with longitudinal data on workers before and after a change in county-level attainment status. He finds that workers in newly regulated plants lost $5.4 billion (in 1990 dollars) in earnings due to the amendments and that these costs were mostly accounted for by a combination of delay in finding a new job elsewhere and lower earnings in future jobs. These losses are substantial, but also quite small relative to the estimated health benefits of the 1990 amendments.

Other researchers have tried to estimate the economic costs associated with this regulation-induced reallocation of production. For example, Greenstone, List, and Syverson (2012) examine the effects of the Clean Air Act on manufacturing total factor productivity. They explore how total factor productivity of polluting establishments in nonattainment counties changes relative to the productivity of polluting firms in attainment counties. They then convert these estimates into foregone output or losses to social welfare. Their estimated loss of total factor productivity in nonattainment counties corresponds to an annual economic cost from the regulation of manufacturing plants of roughly $21 billion, or about 9 percent of manufacturing sector profits in this period. One potential limitation of this study is that it ignores pollution as a factor of production. This may lead to bias when measuring total factor productivity; if a regulation induces firms to use less pollution (i.e. fewer unmeasured inputs) then it may look like total factor productivity declines, when in fact the "true" regulation-induced productivity change remains elusive.

There are clear tradeoffs that researchers face when relying on "program evaluation" methods to answer questions pertaining to the costs of policy. For example, if economic activity is being reallocated from more regulated to less regulated areas, then economic activity in attainment areas will serve as poor counterfactual for nonattainment areas. One potential solution to these problems comes from the growing literature in macroeconomics and international economics that considers problems of how to aggregate difference-in-difference estimates that rely on relative comparisons between potentially linked economic units (see e,g., Nakamura and Steinson 2018, Adao, Arkolakis, and Esposito, 2019).

A more fundamental problem is that the answers obtained from many of these studies are often divorced from economic ideas of efficiency costs or welfare. For example, it is not clear how to interpret findings that plant entry decreases and plant exit increases in response to tighter air quality standards. Other fields within economics may be able to provide useful insights. For example, papers in public finance have considered the welfare and incidence of changes in state corporate tax rates, recognizing that firms may move in response to tax changes. Suarez-Serrato and Zidar

(2016) relate changes in firm entry/exit elasticities to welfare metrics that could prove fruitful in the welfare analysis of clean air regulations. Similarly, the idea that the CAA distorts production decisions is well-appreciated, but researchers have almost no understanding of the allocative efficiency losses associated with these distortions. Fajgelbaum et al. (2016) propose methods to understand how variation in state corporate taxes may lead to allocative inefficiencies in production and economic costs to society, and the same technology may be well suited to shed light on the allocative inefficiencies associated with the CAA.

Relatedly, more work estimating the effects of the CAA on outcomes connected to welfare and incidence (e.g. prices, markups, and marginal costs) would be of tremendous value, as would work that more carefully considers input-output linkages since the latter are likely to affect the overall cost of the Clean Air Act.

4.2 Estimating Compliance Costs Based on Structural Models of Single-Industries

A second approach to estimating the compliance costs of the CAA comes from the "New Empirical Industrial Organization" industry-based studies. These studies focus on a single industry and devote careful attention to institutional details, measurement of key variables, and econometric identification issues. This approach aims to get inside the black box of CAA regulations to understand the mechanisms underlying how they work. The hope is that the research community can learn generalizable insights starting from a relatively narrow focus. However, intra-industry linkages, which may be a significant component of the overall costs, are often willfully ignored.

As one example of this style of research, Ryan (2012) focuses on how the Clean Air Act affected the Portland cement industry. Some relevant features of the industry are high transportation costs and large fixed costs of entry, leading to the possibility of local monopoly. By increasing entry costs, the CAA can exacerbate monopoly power, potentially harming consumers. Using a dynamic oligopoly model, Ryan (2012) estimates that the entry costs created by the Clean Air Act led to multi-billion dollar losses of consumer surplus in this single industry. Clearly, evaluations of the welfare costs of the Clean Air Act should consider possible anti-competitive implications of the policy.

Empirical methods meant to specifically model the details of a single industry have proven particularly useful for understanding how sub-components of the Clean Air Act, such as the Acid Rain Program, have affected the U.S electricity generation industry. By constructing a detailed model of electricity markets, researchers have explored the cost implications of tradable permit markets either relative to a no-regulation counterfactual or compared to conventional command-and-control benchmarks (for example Ellerman et al. 2000; Fowlie 2010; Fowlie and Muller 2013; Chan et al. 2018). Chan et al. (2018) use this approach to estimate the cost savings under Phase II of the Acid Rain program, finding that cost savings from emissions trading are $210-$240 million (in 1995 US dollars) per year. In a review of various analyses, Chan et al. (2012) suggest that sulfur dioxide allowance trading under the Acid Rain Program contributed to cost savings of between 15 to 90 percent compared to conventional performance standards.

While single-industry studies are increasingly being used to understand the regulatory implications of the Clean Air Act on consumers and producers, results may not generalize to other industries/markets. Moreover, most of these models are necessarily partial equilibrium in nature, so that they do not account for effects on other sectors through forward and backward input-output linkages. Busse and Keohane (2007) offer an interesting example of how these linkages can matter; railways transporting coal to electricity generating plants that were subject to Clean Air Act requirements to reduce sulfur dioxide emissions were able to exert market power and increase prices. Lastly, these models have been criticized for relying on strong modeling and equilibrium assumptions that can obscure the link between the underlying data and the estimates.

4.3 Computable General Equilibrium Models

Computable general equilibrium models represent a third way to investigate the compliance costs of the Clean Air Act. These models capture three types of costs that are typically omitted from other models: substitution effects that result from the price changes associated with environmental regulations (for example, the substitution of "clean" for "dirty" goods in consumption, or the substitution of leisure for labor as goods become more expensive); investment effects; and effects on productivity growth. These models also have the advantage that they can be used to estimate a wide variety of counterfactuals (both prospective and retrospective). Ho, Morgenstern, and Shih (2008) review more than a dozen prior US and European analyses based on computable general equilibrium models. As one example, Jorgenson and Wilcoxen (1990) use the IGEM model to estimate that between 1974 and 1985 the mandated abatement costs of the Clean Air Act reduced the real growth rate of GNP by 0.2 percentage points per year, mostly through an increase in the cost of capital. Of course, GNP is not a welfare measure, and thus additional work is necessary to think about the welfare cost associated with this estimate.

The main drawback of these computable general equilibrium models is that they are not transparent and require many untestable assumptions. For example, the IGEM model (Goettle, Ho, Jorgenson, Slesnick, Wilcoxen, 2007), which has been used extensively by the Environmental Protection Agency and other organizations, features over 2,000 equations that jointly define an equilibrium in each period. The complexity arises because these models must fully specify both the demand side and the supply side of the economy, with a full set of demand elasticities and cross-price elasticities for each industry. In practice, these elasticitieswhich are often key to the results of the welfare analysis are imputed or calibrated based on a range of difficult-to-test assumptions. These models often assume full employment, which assumes away regulation-induced unemployment and transition costs. Also, these models have a difficult time incorporating regulation-induced technological change.

While there are obvious potential advantages to using an equilibrium model to explore counterfactual policies, there may be room to improve on the existing work in this area. International trade, a field with a strong intellectual tradition of general equilibrium modeling, has made a number of quantitative advances recently emphasizing both model parsimony and empirical tractability

(See for example, Eaton, Kortum, Neiman, and Romalis (2011), Caliendo and Parro (2015), and Redding 2016). Some of the methods from this literature may prove useful for understanding the economic costs of environmental policy. Shapiro and Walker (2018) borrow insights from this literature to try to better understand the role of environmental policy in explaining the substantial decline in pollution emissions from manufacturing since 1990. They find that virtually all of the observed reduction in pollution emissions can be explained by environmental policy rather than, for example, increases in trade exposure and production offshoring.

4.4 Predicted vs. Actual Costs of the Clean Air Act

One defining feature of the research on the costs of the Clean Air Act is that predicted costs of the regulations are often higher than the costs that actually occur. Morgenstern (2018) provides an overview of the ways in which prospective and retrospective analyses have differed for nine separate environmental policies. There are at least four reasons why this pattern may arise.

First, firms are creative, while models are often parsimonious. It may be difficult to capture the full range of compliance opportunities available to firms in a prospective analysis. Second, and relatedly, unforeseen changes in the economic environment may lead to substantial cost savings. A well-known example is the deregulation of the railroad industry and the sudden opening of access to low-sulfur coal in eastern states (Ellerman et al. 2000). This access to cheap, low sulfur coal was a much more cost-effective compliance strategy for electricity generating units under the Acid Rain Program than either purchasing emissions permits and/or installing scrubbers. This is one possible reason why the actual costs of the Acid Rain Program were substantially lower than initially anticipated.

Third, regulations may increase the return to innovation in abatement, so that the resulting endogenous technical change drives abatement costs lower than earlier forecasts. Popp, Newell, and Jaffe (2010) provide an overview of empirical studies linking pollution abatement costs and expenditures with bursts of innovation as measured by environmental patents. However, credible, causal estimates of induced innovation are difficult to come by. A number of other factors such as greater trade openness and the availability of cheaper imports are also plausible candidates for the observed improvements in compliance technology costs over time.

Fourth, predicted costs of pollution abatement have sometimes failed to anticipate important interactions with existing policies and regulatory regimes. For example, Fowlie (2010) shows how a tradable permit program can be hindered by other regulations that prevent the permit market from reaching the least-cost solution for pollution abatement.

Although in this section we emphasize that forecasters have often over-estimated the costs of environmental policies, researchers have also sometimes under-estimated the benefits. For example, the primary rationale for the Acid Rain Program was the acidification of lakes in the Northeast. The primary anticipated benefit was therefore ecological. However, in the years following the initiation of the program, a new scientific consensus emerged about the harmful impacts of particulate matter on human health. The Acid Rain Program reduced particulate matter along with the targeted SO_2

reductions. Ultimately, over 95 percent of the benefits of the Acid Rain Program were associated with the human health impacts of reduced levels of particulate matter.

5 Distributional Effects

Even if overall costs and benefits balanced precisely, a law with the pervasive scope of the CAA is likely to have profound distributional effects depending on who benefits and who bears the costs. There is a body of evidence showing that poor and minority households are exposed to higher levels of air pollution on average, and these patterns have existed for as far back as measurement has been possible. This fact has inspired a great deal of research on "environmental justice," which is ably surveyed by Banzhaf, Ma, and Timmins (2018) in this journal. If poor and minority persons are more likely to be exposed to pollution, then they may also have benefitted disproportionately from anti-pollution policy. However, the evidence on the distribution of benefits of the Clean Air Act has been somewhat indirect, as researchers have much less information on the spatial distribution of air quality given the sparse pollution monitoring network. For example, fewer than 20 percent of U.S. counties contain a single EPA particulate monitor used to measure compliance with the Clean Air Act (Fowlie, Rubin, and Walker 2019). Researchers have shown that the 1990 Clean Air Act Amendments improved air quality the most in the neighborhoods that surround a violating monitor (Auffhammer et al. 2009), and this has led the benefits of the air quality standards to be progressive in nature (Bento et. al 2015). However, the evidence is necessarily incomplete given existing pollution monitoring network.

Recent improvements in satellite-based measurements of air quality have the potential to relax these existing research constraints. For example, Currie, Voorheis, and Walker (2019) use satellite-based measurements of particulate matter from 2000-2015, connected to Census demographic data to show how recent changes to the Clean Air Acts particulate standard has contributed to disproportionate improvements in air quality for African Americans over this time period. Other parts of the Clean Air Act, such as the elimination of lead in gasoline, have had a disproportionately positive effect on the test scores of minority children (Aizer et al. 2018).

In addition to being more likely to suffer exposure to pollution, poor and minority households may also suffer greater harm conditional on exposure. For example, the effects of air pollution may depend on underlying health; children are more likely to take up lead if they are nutritionally deficient in iron or zinc. The same potential exposure (in terms of, for example, outdoor ambient air quality) could also have a greater impact if disadvantaged people are less able to evade actual exposure; for instance, it may be harder to escape outdoor particulates in a drafty house than in a well-insulated one.

All of these arguments suggest that poor and minority households may have gained the most in terms of health from the targeted nature of the Clean Air Act enforcement; cleaning up the dirtiest regions first disproportionately improved air quality in low-income and minority communities.

While we believe improvements in health are first order, affecting many aspects of day-to-day life, there are additional, possibly second order, distributional issues to be considered. Fullerton and Muehlegger (2017) suggest a number of reasons why the costs of environmental regulation may also be disproportionately borne by disadvantaged households. First, regulation may raise consumer prices and transportation costs, burdening poorer families. Although, robust empirical evidence is lacking. Second, regulation of air pollution may induce firms to substitute to more capital-intensive technologies, thereby reducing demand for unskilled workers (Vona et al. 2018). Third, those who do not own homes or capital will miss out on the economic rents created by higher property values when pollution is reduced (Grainger 2012). Fourth, if cleaner areas gentrify, then lower-income residents, who are more likely to be renters than owners, may end up being pushed out by rising property values.

To summarize, the Clean Air Act is incredibly multifaceted and has most certainly led to significant benefits and costs to different stakeholders over time and across specific policies. To date, the evidence as to these distributional impacts is limited relative to the many possible impacts and mechanisms mentioned above. Going forward, understanding the distribution of the costs and benefits of environmental policy is crucial for policy design and welfare analysis, and may also help shed light on the political economy of pollution controlan area that seems ripe for further study but has attracted relatively little research attention (for discussion of some exceptions, see Oates and Portney 2003).

6 Concluding Thoughts

The CAA is one of the most far reaching pieces of regulatory legislation ever passed in the U.S. Arguably, it affects just about all aspects of daily life, either directly by impacting the air we breathe and the cars we drive, or indirectly by affecting prices and the location of jobs and industries. The CAA is also incredibly complex with many moving parts and both major alterations (especially the amendments of 1977 and 1990) and continuous smaller revisions that may nonetheless have had important effects.

The research literature in this area suggests several broad conclusions. First, the Clean Air Act successfully reduced concentrations of regulated pollutions, although we do not know exactly how much of the spectacular reduction in pollution concentrations in the United States over the past 50 years can be attributed solely to the Clean Air Act. This reduction in pollution has had tangible benefits in terms of peoples health and wellbeing, and people value those benefits. Second, the law has imposed substantial costs, and the costs are considerably greater than direct compliance costs alone. There has been a trend in regulatory policy towards flexible market mechanisms (like permit trading) to achieve compliance while minimizing costs. There are many virtues to these market-based approaches, not the least of which are strong static and dynamic incentives to achieve further emissions reductions that can be lacking in more prescriptive regulatory approaches. Third, there seems to be a general consensus that the benefits of clean air legislation over the past 50 years are likely to have greatly exceeded the costs. That said, it is simply not possible on the basis of the currently available evidence to add up the total benefits and/or the total costs, although the EPA has tried (EPA 2011). One step in that direction would be for researchers to contribute parameter estimates that facilitate "apples to apples" comparisons of benefits or costs across studies (e.g. dollar per ton of pollution reduction). Fourth, it seems that the benefits could have been achieved at far lower cost through more efficient policy choices.

In looking ahead at the near- and medium-term, here are at least four sets of developments that bear particular attention.

First, the Clean Air Act is struggling to come to grips with issues of greenhouse gas emissions. In Massachusetts v. EPA, 549 U.S. 497 (2007), the US Supreme Court had ruled that the EPA has the authority to regulate greenhouse gases. During the Obama administration, the Environmental Protection Agency started a process to begin regulating emissions of greenhouse gases under the Clean Air Act. This decision led to various proposed rules: tighter emission standards for light-duty motor vehicles and certain larger "new and/or modified" stationary sources, as well as a national standard for greenhouse gas emissions from power plants. However, some of these plans have been stayed by the Supreme Court in 2016 while others have been halted by the current administration. The Trump administration has replaced the Clean Power Plan of the Obama administration with its own, less-stringent Affordable Clean Energy rule, which focus on making existing coal power plants more energy-efficient.

Second, the Trump administration has currently rolled back more than 10 regulations designed to protect air quality under the Clean Air Act, with an additional 14 currently being considered. Time will tell as to whether courts agree with the executive branch interpretation of how the law is currently written. Whatever the case may be, these many changes present unique opportunities for empirically minded researchers to learn more about the ways in which these policies have led to measurable gains or losses to society.

Third, there have been dramatic improvements in technology for measuring air quality in recent years. For example, recent advances in satellite technology, combined with advances in prediction techniquesvia machine learninghave allowed researchers to predict ground-level concentrations of PM2.5 at fine spatial and temporal resolutions; e.g. 1km grids on a daily basis (Di et al. (2016)). There are also hundreds of new, low-cost pollution monitors being offered to consumers in efforts to "crowd-source" measurement from the ground up - in real time and with considerable spatial coverage (see e.g. Fowlie 2019). These new ways to measure air pollution with unprecedented speed and granularity have considerable promise both for the design and implementation of Clean Air Act regulation and for research on its consequences.

Finally, while the total benefits of the Clean Air Act appear much greater than the total costs, that does not inform us about the costs and benefits of an additional marginal unit of pollution reduction in air pollution. Said differently, are we currently regulating pollutants in a way that maximizes social welfare where the marginal benefit of a unit reduction in pollution emissions is equal to the marginal costs of abatement for that same unit of emissions reduction? This is one of the central questions in environmental policy, and the answer remains elusive. Our understanding of the benefits of improving air quality are changing rapidly with continued scientific discovery. Estimates of compliance costs are dropping due to ongoing innovations and technological breakthroughs in pollution abatement. These developments offer some exciting starting points for future research.

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