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A Mayor's Perspective on Tackling Air Pollution

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A Mayor's Perspective on Tackling Air Pollution[1]

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Abstract

We review recent empirical economic studies on urban ambient air pollution from a mayor's perspective. We discuss the sources of urban air pollution, the economic costs that it imposes, and the policy tools available to a mayor to alleviate it. For economic costs, we briefly summarize traditional estimates of health and mortality costs and focus on more recent evidence on mental and psychological health, labor productivity and supply, avoidance behavior, willingness to pay for clean air and long-term (multi-decade) impacts. The policy tools we evaluate include pollution information disclosure, auto license and driving restrictions, congestion tolls, public transit investments, emission standards and controls, and gasoline taxes. We also discuss challenges posed by transboundary pollution across cities and the extent to which mayors' incentives encourage tackling air pollution under different political systems. We briefly discuss possible future research agendas.

JEL Codes: H23, H75, O18, Q51, Q52

Keywords: urban air pollution; environmental costs and benefits; urban public policy, environmental policies; incentives

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1. Introduction

Suppose you are the recently elected mayor of Model City, a large city facing severe air pollution. You ran on a platform promising voters that you will improve the city's air quality. How would you fulfill your promise? What are the sources and social costs of air pollution? What policies can you use to reduce the pollution levels your constituents face? How should you decide how much to spend reducing the pollution levels in your city?

First of all, you should know that you are not alone. The World Health Organization (WHO) estimates that in 2012 90% of people that lived in urban areas experienced air pollution that exceeded the WHO's recommended limits and that air quality was generally declining (WHO, 2016). If you are a mayor in a lower-income country then your task is likely even more difficult. Pollution is worst in low- and middle-income countries with 98% of cities not meeting guidelines compared to 56% in high-income countries (WHO, 2016). The consequences of such high levels of pollution are dire. The WHO estimates that in 2012 ambient air pollution caused three million deaths, 87% of which were in low- and middle-income countries, and associated health complications caused 85 million disability-adjusted life years (WHO, 2016).[2]

This chapter's goal is to provide useful advice on these questions for mayors based on high-quality academic economic studies. What do these economic studies offer to help mayors better understand the sources of air pollution, the costs it imposes on cities, and policies that are effective in reducing it? Section 2 describes the local and imported sources of air pollution in cities. Section 3 explores the economic costs that air pollution imposes on cities via its impact on health, mortality, psychological well-being, labor productivity, labor mobility, and out-migration. Section 4 discusses the effectiveness of pollution-reduction policies that cities around the world have implemented – a possible tool box for mayors. Section 5 discusses whether pollution-policy design and implementation is compatible with mayors' incentives including the role of information. The last section summarizes how mayors can use these insights from the studies reviewed in this chapter and proposes future direction for research.

[2] Deaths resulting from particulate matter ($PM_{2.5}$ and PM_{10}) only. Morbidity effects include acute lower respiratory, chronic obstructive pulmonary disease, stroke, ischemic heart disease, and lung cancer.

There are earlier economic review articles on air pollution and cities that take different perspectives. Kahn (2006) reviews the supply and demand of city air pollution and provides a conceptual framework for government policy interventions. Kahn and Walsh (2015) survey theoretical and empirical work on the relationship between environmental amenities (including air pollution) and urban growth. In contrast to these papers, we focus on recent empirical work that measures the economic costs of air pollution and evaluates city-government interventions to reduce these costs. Wherever possible, we focus on papers that provide causal quantification and provide theoretical background only where necessary to interpret the empirical results. We consider articles examining all countries although empirical work thus far has focused predominantly on the US and China.

This is not a comprehensive survey of city air pollution – the scope of which would require a book rather than a chapter. We focus on topics that are prevalent in empirical economics. In doing so, we benefit from a recent surge in empirical studies using careful identification methods and micro data that examine these issues. We impose some boundaries on what we discuss. We look at only ambient, not indoor, air pollution and consider sources largely within a mayor's control. Although we try to include studies on a variety of ambient pollutants, more results relate to particulate matter as it has been studied most extensively and is a common proxy for pollution exposure more generally (WHO, 2016: 19). One gray area that we choose not to cover is power plants. While mayors may influence the location of power plants, this is generally out of their control. Power plants are extremely difficult to move once constructed and the initial location of power plants is based primarily on engineering considerations (see Chen (2021) for a discussion). There is a large literature examining national efforts to reduce sulfur dioxide emissions through allowance markets (Goulder, 2013).

2. Sources of Ambient Air Pollution in Cities

Ambient air pollutants include the six criteria pollutants: particulate matter (PM), sulfur dioxide (SO_2), ozone (O_3), carbon monoxide (CO), nitrogen dioxide (NO_2), and lead.[3] Of these, the most pernicious is typically particulate matter (PM) – small particles that are usually measured as either smaller than ten micrometers in diameter (PM_{10}) or smaller than

[3] Criteria pollutants are the only air pollutants for which the US EPA has established national standards. All of these are directly emitted except for ozone which forms from chemical reactions between nitrogen oxides and volatile organic compounds triggered by sunlight.

2.5 micrometers in diameter ($PM_{2.5}$). Globally, ambient $PM_{2.5}$ in urban areas originates 25% from vehicular traffic, 15% from industrial activities, 20% from household fuel burning, 22% from unspecified human activity, and 18% from natural dust and salt although these compositions differ considerably across cities (Karagulian *et al.*, 2015).

Mayors must also contend with pollution imported from neighboring cities. Although larger particles do not travel as far, $PM_{2.5}$ can travel hundreds of miles making their importation an issue for mayors (Environmental Protection Agency (EPA), 2021). There are a few empirical papers that quantify pollution spillover effects across or within cities. Transboundary air pollution has been shown to significantly affect housing prices across Chinese cities (Zheng, Cao *et al.*, 2014), mortality across Census blocks within Los Angeles county (Anderson, 2019), and manufacturing productivity across major Chinese cities (Fu *et al.*, 2020). Kahn (1999) estimates industry-specific spillovers across US counties of manufacturing activity on total suspended particulates (TSP)[4] and compares them to locally-generated manufacturing pollution. For primary metals manufacturing, which declined the most during the sample period, the elasticity of TSP with respect to local value shipped was 3.5% versus 1.1% for value shipped in an adjacent county. Thus, spillovers are a significant concern for mayors.

3. Economic Impacts of Ambient Urban Air Pollution

Mayors need a comprehensive understanding of the economic costs that air pollution imposes on city residents. Understanding of the scope of pollution's economic costs on cities has broadened recently due to novel empirical economic research. Traditionally, studies focused on two main areas each of which generated a large literature. The first quantified the health and mortality costs of air pollution. The second measured willingness to pay for air quality via its effect on local property values.[5] We will only summarize these two well-developed areas and instead focus primarily on emerging areas. Recent empirical work in economics has quantified new sources of air pollution's costs including effects on mental health, labor productivity, and avoidance behavior. Avoidance behavior, which in the extreme includes out-migration, not only imposes costs but also introduces error in traditional measures of air pollution's costs.

[4] "Total suspended particulate," is an older measure of PM pollution.

[5] Many of the papers in the property values strand used ordinary least squares hedonic models rather than causal estimates of the effects.

Physical Health and Mortality

Short-run exposure to air pollution can lead to decreased lung function, irregular heartbeat, increased respiratory problems, nonfatal heart attacks, and angina while long-run exposure can lead to cardiopulmonary diseases, respiratory infections, and lung cancer (EPA, 2004). The medical literature has a long history of estimating the health and mortality effects of pollution using exposure-response functions. Worldwide, Cohen *et al.* (2017) estimate that $PM_{2.5}$ was the fifth-ranked mortality risk factor in 2015 causing 4.2 million deaths with 59% of them occurring in East and South Asia and 103.1 million disability-adjusted-life-years (DALY). To convert these exposure-response relationships into economic costs, the DALYs can be multiplied by the value of a statistical life (VSL) which is an estimate of the marginal rate of substitution between income and mortality risk (OECD (2012) provides a meta-analysis of VSL estimates). Matus *et al.* (2008) provide a more sophisticated methodology (an integrated assessment model (IAM)) for estimating costs by incorporating these epidemiological exposure-response effects into a computable, general-equilibrium model of the economy. This allows for effects on leisure time and separately estimates medical expenditures as these are resources diverted from other, productive sectors.

These studies rely on a statistical relationship between pollution exposure and health effects. This creates two issues. First, these are not necessarily causal effects due to omitted variable bias, measurement error, and correlations between different pollutants. Deryugina *et al.* (2019) address these issues using high-frequency changes in wind direction as an instrumental variable (proxying for imported pollution). The authors find larger effects on mortality, health care use, and medical costs when instrumenting, consistent with a downward bias in traditional estimates. Cheung *et al.* (2020) use air pollution blowing from mainland China as an instrumental to estimate air pollution's effect on cardio-respiratory mortality in Hong Kong. Second, these studies include the effect of any avoidance behavior and spatial sorting.[6] Therefore, these models generally understate air pollution's true health costs. Recent work on avoidance behavior, summarized below, attempts to quantify this.

[6] People may change residences in order to avoid air pollution. Not correcting for this will result in an under-estimate of pollution's effects. Firms may respond to air pollution by sorting to low-pollution regions (in order to increase productivity – see the "Labor Productivity/Supply" subsection) or to high-pollution regions (the "pollution haven" effect – see Becker and Henderson (2000) and Greenstone (2002)) resulting in either an under- or over-statement of pollution's effects.

Mental and Psychological Health

An emerging empirical literature provides convincing evidence that air pollution adversely affects cognitive and psychological health. Being exposed to air pollution during an exam period or a school year lowered students' contemporaneous test scores in California (Ham *et al.*, 2011), Israel (Ebenstein *et al.*, 2016), and China (Graff Zivin *et al.*, 2019). Long-term cumulative exposure to air pollution reduced students' academic performance (Kweon *et al.*, 2016; Bharadwaj *et al.*, 2017; Heissel *et al.*, 2020), adults' cognitive performance (Zhang *et al.*, 2018), and caused dementia (Bishop *et al.*, 2018). As a countermeasure, installing air filters in classrooms improved students' academic performance (Gilraine, 2020).

Decision-making skills are also adversely affected by air pollution. CO and $PM_{2.5}$ caused US baseball umpires to make more incorrect calls (Archsmith *et al.*, 2018). $PM_{2.5}$ caused UK drivers to have more accidents (Sager, 2019) and degraded quality of political speeches in Ottawa (Heyes *et al.*, 2016b). In an experimental setting, air pollution affected subjects' decision making, increased their risk and ambiguity aversion over gains and made them less prosocial and reciprocal (Chew *et al.*, 2021). These laboratory findings are consistent with investor behavior in financial markets: $PM_{2.5}$ levels lowered New York City investors' returns as measured by a New York Stock Exchange index (Heyes *et al.* 2016a); investors in Chinese stocks performed worse on hazy days (Huang *et al.*, 2020) as do mutual fund investors (Li *et al.*, 2021); and investment analysts in China were more likely to provide pessimistic forecasting on severely polluted days (Dong *et al.*, 2021).

Psychologically, air pollution reduced self-reported happiness or life satisfaction in Germany (Luechinger, 2009), the US (Levinson, 2012), and China (Zhang, Zhang *et al.*, 2017). Extensive public health and medical studies demonstrate that air pollution is associated with annoyance, anxiety, mental disorders, self-harm, and unethical behavior (see review by Lu (2020)). Recently, economists have begun to provide causal evidence of these effects. Air pollution negatively affected self-reported mental health in China (Chen *et al.*, 2018) and increased the rate of depressive symptoms (Zhang, Zhang *et al.*, 2017). Air pollution also increased violent crime in Chicago (Herrnstadt *et al.*, 2021) and London (Bondy *et al.*, 2020).

Labor Productivity and Supply

Historically, pollution reduction efforts have been viewed as purely a tax on city output. Pollution abatement either increases firms' costs per unit of output (e.g., purchasing pollution reduction equipment or hiring compliance personnel) or requires direct reductions in output. However, recent work has shown that air pollution reductions can enhance physical and human capital increasing labor productivity and labor supply. Given labor markets are local, mayoral efforts to reduce local air pollution will bring benefits in increased output and therefore property values and tax revenue that may countervail pollution reduction costs.

The physical and cognitive impairments caused by pollution can reduce labor productivity due to reduced physical or mental effort while at work and death of older, more-experienced workers whom are replaced by younger, less-experienced workers. Labor supply may also fall due to sick days from impaired health of workers or because workers must miss work to care for family members – particularly infants and the elderly who are more vulnerable to air pollution. Empirical studies have quantified both labor productivity (output per hour worked) and labor supply effects. The latter is measured as hours worked, days worked, or number of workers depending on the time frame of the study.

Recent studies measure the relationship between pollution and output carefully, addressing simultaneity and omitted variable biases. Regions with more output will have more pollution leading to a downward bias while if pollution lowers output this will in turn lower pollution leading to an upward bias. In addition, confounding factors may affect both pollution and output, in particular sorting of firms, workers, or regulatory changes in response to pollution. Hanna and Oliva (2015) utilize an exogenous shock (the closing of a single factory in Mexico City) as an instrumental variable to address these endogeneity issues and find an elasticity of -0.18 of hours worked with respect to SO_2 pollution.[7] Graff Zivin and Neidell (2012) estimate an elasticity of output with respect to ozone of -0.26 for California fruit pickers. Since the fruit pickers represent a small fraction of total output, simultaneity bias is not an issue. The authors are able to directly confirm that number of workers and hours worked remained the same so that these are per-hour productivity effects.

Do these effects on productivity extend to indoor workers? This is relevant because $PM_{2.5}$ pollution is small enough that it can permeate buildings in the absence of preventative measures. Chang *et al.* (2016) find significant but more modest effects for indoor pear packers – an elasticity of -0.062 or one-fourth the effects on outdoor fruit pickers. However, He *et al.* (2019) find that cumulative effects may be greater in examining indoor textile

[7] Because a single plant is closed, aggregate goods and labor demand are relatively unaffected. The authors also test for evidence of migration and find none.

workers in China. Cumulative (over 25 to 30 days) exposure to $PM_{2.5}$ results in elasticities ranging from -0.035 to -0.30.

These studies combined with evidence that pollution affects mental and psychological well-being, suggest that knowledge workers may be affected. A number of studies across varying professions find such evidence. Chang *et al.* (2019) find an elasticity of -0.023 in productivity of call center workers in China with respect to the Air Pollution Index (API).[8] These are per-hour productivity effects as the authors find no effect on hours worked or number of workers. These effects extend to high-skilled indoor workers. Kahn and Li (2020) find that decision times of judges in China increased in response to the average Air Quality Index (AQI)[9] over the duration of cases. Meyer and Pagel (2017) find that PM_{10} reduces the likelihood that individual investors in Germany sit down, log in, and trade in their brokerage accounts.

To control for endogeneity, these papers focus on a single firm or type of worker whose output is a small part of aggregate output. While these results are useful for targeted environmental policies, more comprehensive estimates are necessary to evaluate broad-based policies. Doing so also requires moving from partial-equilibrium estimates that ignore the feedback of output on pollution to general equilibrium estimates. Fu *et al.* (2021) address this by identifying causal effects of pollution on output and output on pollution and simulating an IAM to estimate the general-equilibrium effects. The authors find an elasticity of -0.28 of output with respect to $PM_{2.5}$ including both productivity and labor supply effects. The authors also find greater effects for high- than low-skilled workers. Consistent with this, Adhvaryu *et al.* (2019) find pollution affects productivity more for workers performing more complex tasks.

These results suggest that mayors should be aware that air pollution has significant effects on productivity and labor supply and that these effects apply outdoors, to a lesser extent indoors, and to both low- and high-skilled workers. There is much that is still unknown. Little is known about the underlying reasons for reduced productivity. An exception is Aragón *et al.* (2017) which finds that moderate $PM_{2.5}$ levels affect work hours for households with small children and elderly members disproportionately while high levels affect all households equally. The appropriate policy response depends on the underlying cause. For example, if

[8] The API provides a scaled measure of the worst pollutant for each day based on SO_2, NO_2, CO, PM_{10}, and O_3.

[9] The AQI replaced the API in China in 2012. The AQI also provides a scaled measure of the worst pollutant for each day but is based on six pollutants: SO_2, NO_2, CO, $PM_{2.5}$, PM_{10}, and O_3.

lost productivity is due to reduced worker stamina then better workplace air filtration is warranted; while if work is missed to take care of children better school air filtration is in order. More work is needed on the services sector. The evidence for call centers and investors is suggestive but more comprehensive evidence would be useful.

While the firm- and industry-specific studies are able to test for avoidance behavior, the aggregate estimates in Fu *et al.* (2021) are inclusive of avoidance behavior. This could be material. Using detailed task data for Indian garment workers, Adhvaryu *et al.* (2019) show that managers are more likely to reassign workers away from tasks most degraded by pollution. Results so far indicate greater effects for highly-skilled, highly-complex work. More work is needed to confirm this but, if so, there is an added urgency for mayors to reduce pollution because these workers contribute disproportionately to output.

Avoidance Behavior

Understanding avoidance behavior is important for mayors for three reasons: increased information disclosure may lead to defensive actions that better protect citizens, estimates of pollution's costs that do not account for avoidance behavior may be understated, and the most extreme form of avoidance behavior, migration, may reduce the city's productive work force and long-run growth.

Pollution information disclosures such as smog alerts can help people take actions to reduce pollution exposure. Even absent public air quality information, people can identify, with error, pollution severity based on visibility, haze, or respiratory reactions. There are three types of avoidance behavior: reducing outdoor activities; buying protective equipment; and traveling or moving to locations with better air quality.

When air pollution is moderate (AQI between 200 and 300), elderly and people with respiratory diseases are recommended to stay indoors while when it is above 300 even healthy people are recommended to do so. Research indicates many people follow these guidelines at least partially. In Southern California, the first day of a smog alert caused attendance at a public zoo and an observatory to drop by 15% and 8% respectively; however, decreased attendance diminished dramatically on the second consecutive alert day and disappeared on the third (Graff Zivin and Neidell, 2009). Children and the elderly responded more to alerts than non-elderly adults (Graff Zivin and Neidell, 2009; Neidell, 2009). In Texas, school absences were more likely during bad air pollution, which could be due to students becoming sick or parents keeping them home to avoid exposure (Currie *et al.*, 2009). In China, international students (Liu and Salvo, 2018) and Chinese students in big cities (Chen *et al.*, 2018) were more likely to be absent on polluting days and movie attendance declined with pollution (He *et al.*, 2020).

Purchasing defensive equipment is another way to avoid pollution. Chinese households' expenditures on facemasks increased by 55% when air quality moved from slightly to heavily polluted (AQI increase of 100) (Zhang and Mu, 2018). Runners even wore facemasks during the 34th Beijing International Marathon held in 2014 because of severe pollution on the race day (Guo and Fu, 2019). People buy home-use air purifiers to reduce indoor air pollution (Ito and Zhang, 2020). High-income households are more likely to do so (Sun *et al.*, 2017) suggesting income inequality may play a role in defensive expenditures.

The most extreme form of avoidance behavior is traveling or migration. On severe pollution days, online search frequency for international emigration in China increased (Qin and Zhu, 2018). Chen *et al.* (2020a) find that between March 2008 and April 2010, a degradation in Beijing's air quality relative to another city increased air travel from Beijing to that city. The increase was greater for first-class flyers again suggesting income inequality. Similarly, cell phone data for 25 Chinese cities in 2016 indicates that the difference in air quality between two cities led to increased population flow to the city with cleaner air (Chen *et al.*, 2020b). Over a longer horizon, people can relocate to cities with better air quality. Using four waves of population census data in China from 1996 to 2010, Chen *et al.* (2017) find that a 10% increase in $PM_{2.5}$ in a county causes 27 people per 1000 inhabitants to move out with greater effects for college-educated residents.

These empirical results suggest that citizens engage in significant avoidance behavior and defensive expenditures. This is good news in that pollution damage is mitigated although there is some evidence that poor citizens are at a disadvantage in defensive investments. However, it is bad news in that estimated damages from pollution are understated to the extent this avoidance behavior is unaccounted for. Mayors should be most concerned about the evidence on permanent migration. High pollution levels may reduce their labor force and tax base with greater effects for high-skilled workers, which are the main driver of long-run urban growth (Shapiro, 2006). More evidence is needed on long-run avoidance behavior and the inequality in avoidance behavior.

Willingness to Pay for Air Quality

Residents' willingness to pay for air quality is important for a mayor to know so that it can be compared to pollution reduction costs. However, as air quality is a non-market good, marginal willingness to pay (MWTP) for air quality must be deduced through indirect methods. Although there are many approaches,[10] we focus on the three approaches most intuitively understandable to mayors.

The first regresses self-reported happiness or life satisfaction ratings on individual characteristics including income, air quality, and residential location. The MWTP for air quality is inferred from the marginal rate of substitution between income and air quality that keeps individuals equally happy. Using the US General Social Survey data from 1984 to 1996, Levinson (2012) estimates a MWTP of $459 to $891 for a one $\mu g/m^3$ annual reduction in PM_{10}. Luechinger (2009) uses the installation of power plant scrubbers and wind direction as an instrument to control for endogeneity and estimates a MWTP of EUR 313 for a one $\mu g/m^3$ annual reduction in SO_2 using the German Socio-Economic Panel data from 1985 to 2003.

The second approach is the quality of life literature which utilizes hedonic models. Good air quality is an urban amenity specific to a location. As more people move to a location with good air quality, housing demand there will increase and MWTP for air quality will be reflected in the increased housing costs. That is, air quality is capitalized into property values. At the same time, workers may tolerate lower wages in locations with good air quality. Thus, housing and wage hedonic models together, controlling for a set of urban amenity variables, can recover MWTP for air quality (Blomquist, 2006). Chay and Greenstone (2005) use non-attainment status under the US Clean Air Act as an instrumental variable to estimate an elasticity of -0.20 to -0.32 for median property values with respect to TSP using data from 1972 to 1983 (a one $\mu g/m^3$ decrease in TSP increases property values by 240 in 2001 USD). Using data for 85 Chinese cities from 2006 to 2009 and imported pollution as instrument, Zheng, Cao *et al.* (2014) estimate a much smaller elasticity: -0.08 for home prices with respect to PM_{10}. Bayer *et al.* (2009) incorporate moving costs in a locational discrete choice model and find hedonic models that ignore moving cost under-estimate people's MWTP for air quality. The authors estimate that the median household is willing to pay USD 149 – 185

[10] Other approaches include the contingent valuation method, stated preference method, and recreation demand model.

(in 1982-84 USD) for a one μg/m^3 decrease in PM_{10}, – three times larger than estimates from hedonic models.[11] Causal estimates of MWTP in wages are lacking perhaps because the quality of life literature emerged in urban rather than environmental economics. Since recent work has shown that air pollution may affect labor productivity and labor supply both directly and through migration, wage hedonic estimates would need to be adjusted for these.

The third approach uses expenditures to reduce pollution exposure or treat its effects. For non-marginal air quality improvements, these can approximate willingness to pay (WTP) for air quality (Bartik, 1988). Zhang and Mu (2018) estimate that facemask expenditures in China increased CNY 610 thousand per severely polluted day (AQI above 300) during 2013 to 2014. Ito and Zhang (2020) estimate that a Chinese household is willing to pay USD 1.34 annually to remove one μg/m^3 of PM_{10} based on scanner data for air purifier sales. Deschênes *et al.* (2017) estimate that the Nitrogen Oxides Budget Program in the US from 2003 to 2008 not only reduced air pollution but also pharmaceutical expenditures to address respiratory and cardiovascular problems by USD 800 million (1.6%) in participating states. The authors estimate this represents over one-third of overall WTP for pollution reductions. Using data tracking asthmatic's use of rescue medication, Williams *et al.* (2019) estimate that a 4.5 μg/m^3 increase in $PM_{2.5}$ is associated with a 3.6% increase in rescue medication use. Converting this to WTP and extrapolating to all US asthmatics, a one μg/m^3 reduction in $PM_{2.5}$ nationwide would save USD 350 million annually.

These estimates provide mayors with a few estimates of WTP for air quality but primarily for the US and China. More work is needed on estimates for other countries. Most notably lacking are causal estimates for effects on wages.

Long-Term Impacts

Air pollution can have very long-term impacts on individuals who were exposed early in life. Isen *et al.* (2017) find that US cohorts born in years with a higher pollution level tend to have lower labor force participation and earnings at age 30. In China, childhood exposure to higher air pollution causes fewer schooling years and lower earned income in adulthood (Ebenstein and Greenstone, 2021). Air pollution can also have very long-term impacts on neighborhood stratification and land use due to spatial sorting of city residents. Lin (2018) finds that US census tracts located downwind of 1970 industrial sites have lower housing prices, lower

[11] Freeman *et al.* (2019) apply the same method in China.

shares of skilled employment, and lower wages in 2000; and they also have lower growth rates in all these outcomes from 1980 to 2000. Heblich *et al.* (2021) geocode industrial chimneys in 70 English cities as of 1880 and recreate the spatial distribution of air pollution at that time using an atmospheric dispersion model. Using the 1881 census data, the authors find that highly-polluted locations attract a higher share of low-skilled workers. Most striking, a location's pollution level in 1880 increases the share of low-skilled workers in 1971, 1981, 1991, 2001, and 2011 suggesting very persistent effects. The authors show that once a neighborhood passes a pollution threshold, it develops low amenities and continues to attract low-skilled, low-income residents long after the historical pollution has waned.

These studies suggest that a mayor's response to air pollution can reverberate for decades and very high levels of pollution may permanently trap some parts of the city in disadvantage.

4. Policies to Alleviate Ambient Urban Air Pollution

Information Disclosure

Monitoring and announcing real-time and future projected air quality information is helpful for residents to make informed decisions about avoidance behavior. The US EPA issues Air Quality Alerts at times when ground-level ozone or particle concentrations reach, or are approaching, unhealthy levels in an area and also in the late afternoon when either is predicted to be elevated on the following day.[12] People respond to these alerts by taking actions to avoid exposure such as reducing outdoor activities (Graff Zivin and Neidell, 2009; Neidell, 2009). Awareness of severe pollution may lead residents to buy facemasks and air filters to protect themselves or show a greater interest in relocation (see the Avoidance Behavior section). Another potential upside of information disclosure is that citizens may take actions to reduce pollution on severe days such as by taking public transit instead of driving (Cutter and Neidell, 2009). Disclosure is also important for timing large gatherings such as sports events when air quality is good.

Although most cities in developed countries monitor and publicize daily and even hourly air quality information, many cities in developing countries lag behind. Chinese cities started to roll out automated monitoring stations in 2012 for both PM_{10} and $PM_{2.5}$ (previously only PM_{10} data were collected and only manually). Greenstone *et al.* (2020) find that the PM_{10}

[12] Description is available at https://www3.epa.gov/region1/airquality/smogalrt.html (accessed on April 14, 2021).

concentrations reported by the automated monitoring network were significantly higher than previous readings suggesting that automation significantly reduced underreporting, data manipulation, or both. The introduction of automated collection also increased online searches for face masks and air filters when air quality was bad, suggesting that more precise information and better access helped residents make more informed decisions.

In the absence of $PM_{2.5}$ concentration information, urban residents in China are often confused as to whether severe pollution is fog or smog since they must judge based on visibility. Barwick *et al.* (2019) find that $PM_{2.5}$ disclosure increased media coverage about air pollution, increased defensive expenditures on air purifiers, and reduced outdoor activities on highly-polluted days with a resulting reduction in pollution-related mortality. Interestingly, the housing-price discount of pollution also increased. Importantly for mayors, the authors conclude that improving access to air pollution information is a low-cost, high-return policy for alleviating pollution's effects.

Auto License and Driving Restrictions

While not a new idea – it goes back to at least the times of ancient Rome (Matthews, 1960) – driving restrictions have been increasingly used across the world as urban air pollution has deteriorated. These policies usually restrict cars from driving one or more days per week during certain hours based on the last digit of their license plate number. This is one of the few policies that may address road dust, which contributes to $PM_{2.5}$.[13] Driving restrictions will not necessarily reduce pollution due to purchases of second vehicles, inter-temporal substitution to non-restricted periods, and substitution to dirtier forms of transport (Zhang, Lin *et al.*, 2017). There is probably no other city-level policy that has more mixed evidence than driving restrictions.

The first systematic and rigorous analysis of driving restrictions (Davis, 2008) finds no improvement in air quality from Mexico City's *Hoy No Circula* policy implemented in 1989 due to an increase in the number of vehicles on the road, especially higher-emissions used vehicles. Consistent with this, Gallego *et al.* (2013) use hourly CO emissions as a proxy for vehicle use and find more vehicles on the road in response to the policy. Blackman *et al.* (2018) use a contingent valuation approach to estimate the cost of the *Hoy No Circula* program in 2013 at USD 130 per vehicle per year conditional on vehicle and location choice.

[13] Road-watering policies also reduce road dust but we know of no economics research on these.

However, these policies have been found to be effective in other contexts. Viard and Fu (2015) find that Beijing's driving restrictions, implemented in 2008 improved air quality, perhaps because China's rapid growth meant that few used cars were available. However, the policy also reduced work time for those with discretionary work hours. Consistent with reduced car usage in response to the policy, Xu *et al.* (2015) find that housing prices increased near subway stations and relatively more for those with better commute times. Zhong *et al.* (2017) exploit variation in the number of vehicles restricted each day due to superstitions in China about the last digit on license plate numbers. The authors find that days with more allowed vehicles have higher pollution levels and more ambulance calls. Blackman *et al.* (2020) find that the Beijing restrictions imposed costs of USD 54 to 107 per vehicle per year which are below the benefits of pollution reduction identified in Viard and Fu (2015).

In contrast to the mixed evidence for vintage-agnostic restrictions, studies have found vintage-specific restrictions to generally be effective. Theoretically, such restrictions may induce drivers to upgrade to cleaner vehicles but may also induce drivers of non-qualifying vehicles to travel further to avoid the zones so that the effects are indeterminate. Santiago's vintage-specific driving restrictions implemented in 1992 were effective in reducing pollution because they induced drivers to adopt cleaner vehicles (Barahona *et al.*, 2020). Wolff (2014) finds that Germany's Low Emissions Zones (LEZs), allowing only vehicles with low PM_{10}-emissions in certain zones, are effective at lowering pollution. These German LEZs are also effective in reducing PM_{10} across a broader set of zones and cities (Malina and Scheffler, 2015), improving infant health (Gehrsitz, 2017), and reducing cardiovascular disease (Margaryan, 2021). However, Bento *et al.* (2014) find that allowing single-occupant, low-emissions vehicles into high-occupancy vehicle lanes lowers welfare due to increased congestion for carpoolers.

Episodic driving restrictions are sometimes used to address acute, temporary pollution spikes. These are less likely to result in long-run avoidance behavior such as purchasing a second vehicle. deGrange and Troncoso (2011) find that temporary, rush-hour bans on all cars decreased pollution and increased subway but not bus ridership. Han *et al.* (2020) investigate a temporary, sixteen-day driving restriction policy in Jinan in 2009 and find significant reductions in CO and PM_{10}.

An alternative policy for reducing the number of cars on the road is restricting automobile licenses. Their use has increased in China as the stock of cars has risen along with increased wealth. The two primary methods have been auctions and lotteries although some cities use hybrids (Xiao, *et al.*, 2017). Using the random outcomes to compare winners and losers, Yang, Lin *et al.* (2020) find that Beijing's lottery reduced the total stock of cars by 14% and vehicle kilometers traveled by 15%.[14] Using a structural model, Xiao *et al.* (2017) finds that the restrictions from Shanghai's auction increased welfare because the benefit from reduced externalities exceeded the loss from reduced vehicle transactions given reasonable assumptions about vehicle life, license prices, and externalities. Li (2018) compares the welfare effects of Beijing's lottery to the counterfactual of a uniform-price auction similar to Shanghai's. While the lottery is more effective at reducing automobile externalities (because externalities increase in willingness-to-pay for a car), overall welfare is much higher under the auction because of the allocative inefficiencies in car usage under the lottery.

Barahona *et al.* (2020) construct a structural model that allows for vintage-specific driving restrictions with a uniform program as a limiting case. Using data from a Santiago driving restrictions program that exempted cars with catalytic converters beginning in 1992, the authors show that vintage-based restrictions that affect choice of cars driven (extensive margin) are preferable to uniform restrictions that affect car usage (intensive margin). Placing more onerous limits on older, dirtier vehicles encourages drivers to upgrade to newer, cleaner vehicles in contrast to uniform restrictions which encourage adoption of a second (possibly dirtier) vehicle.

Theoretically, scrappage programs induce car owners to replace older, dirtier vehicles with newer, cleaner vehicles. However, empirical evidence has generally found them to be ineffective due to adverse selection. Li *et al.* (2013) find that a one-month program in the US in 2009 resulted in limited pollution reductions in part because 45% of the subsidies went to consumers who would have purchased a new vehicle absent the program. Similarly, Sandler (2012) finds that in a long-running California program (from 1996 to 2010) owners are more likely to scrap vehicles with few remaining miles to be traveled and therefore a short time for pollution production. Jacobsen and van Benthem (2015) quantify the reduced effectiveness of

[14] These are partial equilibrium effects – holding congestion and other factors affecting car adoption fixed at the time of the lottery.

these programs due to the resulting increase in used car prices estimating a scrappage elasticity of -0.7 with respect to used vehicle prices for the US between 1993 and 2009.

Barahona *et al.* (2020) extend their analysis to compare driving restrictions to license-restriction and scrappage policies. Vintage-based driving restrictions compare favorably to scrappage policies because the latter does not allow older cars to relocate to less-polluted settings where they remain valuable. A gasoline tax (discussed below) outperforms vintage-based restrictions in the short run by deterring car usage but is disadvantageous in the long run as it does not induce a move toward cleaner (per-gallon) vehicles. Vintage-based registration fees outperform vintage-based driving restrictions because they allow a full menu of prices that reflect the marginal social cost of using each vintage.

What do these results mean for mayors? Vintage-specific registration fees are likely the most effective means to reduce auto emissions but may be infeasible to implement. If infeasible, vintage-specific driving restrictions would be next-most effective and would be preferable to scrappage programs which suffer from unintended consequences.

Congestion Tolls

Congestion tolls have been extensively discussed theoretically and have been implemented in several cities. Although the primary goal of these tools is to reduce traffic congestion, as a byproduct they reduce air pollution (Parry *et al.*, 2007). As a mayor, how should a congestion toll be calculated and is it practically feasible to levy these tolls in urban areas? If so, how effective are they in reducing pollution?

When traffic on a road exceeds its capacity, congestion occurs and each additional driver slows down all other drivers increasing their travel costs of gasoline, vehicle maintenance, and opportunity cost of time. The marginal increase in all others' travel costs is the "congestion externality" imposed by the additional driver. The optimal toll charged to each driver should be set to equal to this congestion externality to ensure socially-optimal usage of the road (Arnott and Kraus, 2003). Incorporating the pollution externality into this calculation requires estimating the increased pollution that occurs from all vehicles when an additional driver joins the road. Since congestion varies across time (e.g., rush versus non-rush hours) and roads (e.g., downtown versus suburban areas), optimal congestion tolls should be time- and road-specific (Arnott *et al.*, 1993).

Historically, setting and implementing optimal congestion tolls was difficult. Setting a congestion toll even approximately close to the optimal, requires real-time data on traffic flows. This is only recently feasible with the advent of sophisticated communication and GPS technologies. Moreover, in the past collecting tolls required stopping vehicles to collect physical currency. Recent technologies, such as “smart” cards that are installed in vehicles and scanned while passing a toll collection point, have automated this process.

Although technological hurdles for implementing congestion tolls have diminished, mayors still face political, economic, and social constraints that limit their effectiveness in reducing pollution. Congestion tolls are regressive and may be opposed on this basis. Drivers may avoid tolled areas by driving around them and increase pollution. Drivers may be concerned about the privacy of their travels and avoid using tolls or oppose them politically. Although congestion tolls have not yet been widely adopted, Singapore, London, Stockholm, Milan, San Diego, Houston, Toronto, Seoul, and some Norwegian cities have implemented them (Small and Verhoef, 2007; Lindsey *et al.*, 2008; Anas and Lindsey, 2011).

For mayors, what do economic studies have to say about congestion tolls and how they influence pollution? A few papers have estimated optimal congestion tolls. The main difficulty in doing so is obtaining an exogenous shift in traffic density to identify the increased travel costs. A good example of how to overcome this is Yang, Purevjav *et al.* (2020) who use exogenous shifts in daily traffic density due to plate-number rotations under Beijing’s driving restrictions (some digits are favored over others and therefore in greater use). The authors estimate optimal congestion tolls of CNY 0.15 per vehicle-kilometer for peak hours and CNY 0.10 for off-peak hours using 2014 data.

We are not aware of any empirical estimates of optimal congestion tolls inclusive of the pollution externality. However, there are a few empirical studies that estimate how imposing congestion tolls affect pollution. Using a differences-in-differences (DD) approach with other UK cities as a control group, Green *et al.* (2020) find that London’s congestion pricing program implemented in 2003 reduced PM_{10} by 5.6 to 7.7% and CO by 6 to 9 % depending on the controls employed but increased NO_2 increases by 14 to 17%.[15] NO_2 may have increased because diesel vehicles (buses and taxis), which produce more NO_2, were exempted and increased their driving due to reduced congestion brought about by the tolls. Simeonova

[15] The program, introduced in 2003, charged GBP 5 initially, increasing to GBP 8 in 2005 and GBP 10 in 2011.

et al. (2019) use a DD approach with other Swedish cities as a control group to examine Stockholm's congestion pricing program implemented in 2007.[16] The authors find that the toll reduced PM_{10} by 10 to 15% depending on the controls employed and NO_2 by 15 to 20%. Milan suspended its congestion pricing program for about two months in mid-2012 due to a lawsuit.[17] Gibson and Carnovale (2015) take advantage of this unexpected policy change and estimate the suspension increased CO by 6% and PM_{10} by 17%.

Chinese cities charge highway tolls to finance road construction rather than to reduce congestion (Beijing and Guangzhou are currently discussing implementation of the first urban-center congestion programs in China). Beginning October 1, 2012, the central government waved highway tolls on four nationwide holidays. Fu and Gu (2017) use a regression discontinuity (RD) design combined with a DD estimate using the previous years as a control group to estimate pollution effects during the first National Day holiday to be exempted (October 1 to 7, 2012). The authors find that air pollution (predominately PM_{10}) increased by 20%. Based on average toll data, the elasticity of urban air pollution with respect to tolls is -0.15.

Economic studies confirm that congestion tolls can be effective in reducing auto pollution. However, they do not offer much guidance on setting the optimal toll that takes account of pollution externalities. Future work on this would be useful. In implementing congestion tolls, mayors must consider the practical technological, political, and social constraints.

Public Transit Infrastructure

Many cities invest in public transit (subway, light rail, bus rapid transit) in an effort to reduce congestion and pollution. Theoretically, public transit may or may not be effective in doing so. On the one hand, drivers may substitute to public transit and reduce vehicle kilometers traveled (Adler and van Ommeren, 2016; Bento *et al.*, 2005; Liu and Li, 2020). On the other hand, if latent demand for automobile trips exists, reduced road congestion from expanded public transit can be offset by new drivers – the "fundamental law of highway congestion" (Duranton and Turner, 2011). In addition, urban sprawl can reduce public transit ridership (Baum-Snow *et al.*, 2005). Whether public transit reduces pollution is therefore an empirical matter. The small amount of evidence thus far indicates that public transit is effective in

[16] The toll varied by time of day but did not exceed USD 2.60.
[17] Before this the charge was EUR 5.

reducing pollution for heavily polluted cities but the effects are highly local.

Chen and Whalley (2012) use hourly pollution data and an RD design to estimate the pollution effect from opening the Taipei Metro's first line in 1996. The authors find a 5 to 15% reduction, depending on the specification, in CO but little effect on ozone. Between 2008 and 2016, Beijing built fourteen subway lines and 252 stations. Using this setting and historical subway planning as an instrumental variable, Li *et al.* (2019) find that a one standard deviation increase in subway network density improved city air quality by 2% and that air quality within two kilometers of a new subway line improved by 7.7% relative to areas more than twenty kilometers away suggesting a highly local effect. Bauernschuster *et al.* (2017) use DD estimates applied to 71 one-day strikes in the public transportation sector in five German cities between 2002 and 2011. The authors find that the absence of public transit during morning rush hours increased PM_{10} by 14% and NO_2 by 4%.

Gendron-Carrier *et al.* (2021) provide more comprehensive evidence using a large sample of subway stations from 58 world cities that opened between August 2001 and July 2016. The authors employ an event study based on monthly pollution data before and after subway openings and find no average effect on $PM_{2.5}$. However, there is substantial heterogeneity: $PM_{2.5}$ fell in 26 cities, increased in 20, and did not change in 12. The cities with declines are overwhelmingly above the median in initial pollution levels and $PM_{2.5}$ fell by 4% for all cities above the median. Similar to Li *et al.* (2019), the authors find larger pollution reductions near city centers where subway ridership is concentrated and little effect beyond 25 kilometers. Gu *et al.* (2021) provide complementary evidence for 45 newly-opened subway lines in 25 Chinese cities between August 2016 and December 2017. Opening lines increases rush hour speed by 4% on roads nearby subway lines with effects declining in distance from the lines.

Empirical evidence so far suggests that public transit likely reduces air pollution in locations that have high population density, are close to the access points, have high pollution levels, and induce little automobile trip demand. Improving access to public transit to ensure ridership is also important. For example, many Chinese cities have introduced shared bicycles to help solve the "last mile problem."[18]

[18] For example, see "Bike-Sharing Data and Cities: Lessons from China's Experience," Global Environment Facility, January 17, 2018 available at https://www.thegef.org/blog/bike-sharing-data-and-cities-lessons-chinas-experience (accessed on April 14, 2021).

Emissions Standards and Controls

Emissions standards are generally set at the supra-city level but implemented at the local level and therefore relevant for a mayor. Economic studies have found emissions targets to be effective locally in both the US and China. The studies generally do not identify the mechanisms used to reduce pollution but show that the targets themselves are useful, at least if set in the right way. The US Clean Air Act implemented in 1990 provides evidence that emission standards are an effective city-level tool for reducing pollution. Although a federal law, US counties were responsible for implementation and those in non-compliance faced sanctions. Bento *et al.* (2015) show that PM_{10} pollution reductions from the Act were highly localized by examining changes in housing prices in close proximity to a monitoring station (all monitors had to be in compliance to avoid sanctions).

A similar policy in China – the Air Pollution Prevention and Control Law – has been shown to be effective in reducing air pollution. One iteration of this law implemented in 1998 as the Two Control Zone (TCZ) policy designated 175 of 333 prefectures that exceeded nationally-mandated thresholds for SO_2 and faced more stringent regulations.[19] Tanaka (2015) finds that the TCZ policy successfully reduced infant mortality in prefectures subject to it relative to those that were not.

Mayors must be careful in setting emissions targets to avoid incentive misalignment. The US Clean Air Act Amendment of 1977 specified that a county was in attainment as long as the highest hourly reading over all hours and days of the year did not exceed a certain limit.[20] In response, local regulators relocated polluting industries from more- to less-polluted counties to avoid triggering non-attainment in the dirtier counties (Henderson, 1996) and shifted pollution from monitored to non-monitored days (Zou, 2021). While not necessarily detrimental to improving air pollution, these actions were inconsistent with the law's intention which was to reduce source emissions.

Vehicle smog checks are a tool that mayors may use although Type I and Type II errors have both been shown to plague them.[21] Oliva (2015) finds that cheating is widespread in Mexico City through the use of substitute "donor" cars to pass the test for vehicles that would

[19] Although the law imposed some specific measures that must be taken by non-compliant prefectures, local regulators had significant discretion over how to meet the targets.
[20] The first day with the highest annual hourly reading was exempted.
[21] A Type I error is a "false positive" ("an innocent person is convicted") and a Type II error is a "false negative" ("a guilty person is not convicted").

otherwise fail. As the author points out requiring pollution-reduction equipment on new vehicles is easier to enforce given the smaller number of manufacturers to monitor.[22] Hubbard (1998) finds evidence of Type I errors in California's smog checks – inspectors use their discretion to avoid passing vehicles in order to sell repairs for failed vehicles. Sanders and Sandler (2020) find that California emissions checks were effective in reducing ambient air pollution from older but not newer vehicles based on numbers of re-inspections after failures by each. This suggests that improvements in engine technology over time may render these tests less effective.

Both general emissions standards and smog checks appear to be useful tools that mayors can use to reduce pollution but they must be cognizant of incentive alignment and unintended consequences for both.

Gasoline Taxes

Theoretically, a mayor could employ gasoline taxes as they have been shown effective in reducing auto emissions (Fullerton and Li, 2005). However, implementing gas taxes is not a viable option for most cities except large or geographically isolated ones that can prevent diversion of gas purchases to neighboring cities. This is consistent with the empirical evidence. Only one out of five of the largest cities in each US state have either an excise tax or a sales tax on gasoline or both (Michael, 2017).[23] Presumably, smaller cities would be even less likely to have a gasoline tax. In China, there is a separate fuel tax although, consistent with preventing arbitrage across cities, it is a uniform national rate.

Transboundary Air Pollution

Pollution that drifts from neighboring cities is outside of the mayor's direct control. These externalities can be internalized at a higher political level by exerting centralized control as shown by Yang and Chou (2018). The US Clean Air Act allows for such a procedure. Its Section 126 allows a downwind state to petition the federal-level EPA to take action against an upwind state that impedes its ability to comply with pollution standards.[24] Transboundary pollution can also be resolved through negotiations between two or more cities although this

[22] Although the Volkswagen emissions cheating scandal in 2015 shows that this is not foolproof either "Volkswagen: The Scandal Explained," *BBC News*, December 10, 2015.
[23] Sales taxes on gasoline do not necessarily differ from sales taxes on other items and therefore are not specifically targeted at reducing vehicular miles traveled and therefore pollution.
[24] This is described at https://www.epa.gov/ground-level-ozone-pollution/ozone-national-ambient-air-quality-standards-naaqs-section-126 (accessed on April 14, 2021).

requires clear assignment of property rights (Coase, 1960). For example, in 2012 Hong Kong SAR and Guangdong province in China agreed on joint pollution-reduction targets for the region.[25] As the central government was not involved in the negotiations of this agreement, this is an interesting example of Coasian bargaining at work.[26]

Regardless of the approach used, a quantification of imported pollution's costs as a function of distance is required for the higher political authority to set damages or for cities to negotiate prices between themselves. Chemical-transport models (CTMs) quantify imported pollution via simulation (e.g., Seigneur and Dennis, 2011) but not its costs upon arrival. These would need to be combined with causal damage estimates. Fu *et al.* (2020) provide a method for estimating costs as a function of distance using daily pollution and weather data.

In the absence of such solutions, neighboring mayors are caught in a Prisoner's Dilemma in which there is an incentive to produce socially-excessive levels of pollution and to locate pollution sources close to and upwind of adjacent cities – phenomena that have been documented empirically (Helland and Whitford, 2003; Bošković, 2015). Such free-riding can sometimes be subtle. For example, electric vehicle subsidies initiated by one city can increase air pollution in other cities through increased electric generation (Holland *et al.*, 2016).

Incentives

US mayors' incentives revolve around the electoral process. In the short run, voters exert pressure via whether a mayor gets re-elected (see evidence in List and Sturm (2006) for state governors). In the long run, they subject mayors to inter-jurisdictional competition and Tiebout sorting (Tiebout, 1956). Mayors face a tradeoff which can lead to either a "race to the bottom" or a "race to the top." On the one hand, they have an incentive to attract capital (inter-jurisdictional competition) to generate more local economic activity which can cause inefficiently high pollution levels (Oates and Schwab, 1988). On the other hand, cities compete in urban amenities and quality of life, including air quality, which gives mayors an incentive to reduce inefficiently high pollution levels. This Tiebout sorting arises from residents' ability to move (i.e., "vote with their feet"). Which effect dominates depends crucially on the applicability of these models' assumptions.

[25] "A Clean Air Plan for Hong Kong," Environment Bureau of Hong Kong, March 2013.
[26] See "Legislative Council Panel on Environmental Affairs: Report on the Cleaner Production Partnership Programme," Legislative Council Paper No. CB(1)869/19-20(01) available at https://www.legco.gov.hk/yr19-20/english/panels/ea/papers/eacb1-869-1-e.pdf (accessed on April 14, 2021).

Millimet (2014) summarizes these theoretical models and the empirical evidence concerning their underlying assumptions and concludes that the evidence is not yet conclusive as to whether there is a race to the top or bottom. A few papers directly test how moving from a centralized to a decentralized system of governance affects pollution levels. The results are consistent with a race to the top or at least the avoidance of a race to the bottom. List and Gerking (2000) find that environmental quality either continues to improve or did not decline after US environmental regulation was decentralized to the states in the early 1980s while Millimet (2003) finds no significant change using different econometric techniques. As Millimet (2014) notes these results do not necessarily mean that the decentralized outcome is more efficient than the centralized.

China's central government employs a "tournament competition" to promote local government officials. Before 2005, the promotion criteria were based mainly on local GDP growth (Li and Zhou, 2005; Yu *et al.*, 2016) giving local officials an incentive to sacrifice environmental quality for growth (Wu *et al.*, 2013; Jia, 2017). In 2005, environmental quality and protection were added to the promotion criteria including reducing SO_2 emissions in targeted cities through the TCZ program. Chen *et al.* (2018) use a DD approach with non-TCZ cities as a control and find that cities subjected to TCZ reduced SO_2 emissions more but at the cost of reduced GDP growth. Consistent with this, city mayors and party secretaries whose regions reduced pollution more were more likely to be promoted (Zheng, Kahn *et al.*, 2014; Wu and Cao, 2021).

An unintended consequence of environmental-based performance evaluation is that local officials have an incentive to manipulate environmental data. Many Chinese cities are required to reach the goal of 85% "blue-sky days" (API less than 100) in a year. Ghanem and Zhang (2014) find evidence of sharp discontinuities at the blue-sky day threshold for 50% of the 113 cities in their data and that such manipulation is more likely to occur on days with high visibility when manipulation is hardest to detect. Such manipulation is also effective as it is correlated with future promotions of officials (Ghanem *et al.*, 2020). Nonetheless, the API is highly correlated with two alternative measures of air pollution: visibility and Aerosol Optical Depth, suggesting that the API contains useful information (Chen *et al.*, 2012).

5. Conclusion

Recently, high-quality empirical economic research that is relevant for mayors in tackling air pollution has blossomed. However, because the baseline of research that examines "micro" issues of policy relevance for mayors was previously small there remains much work to be done. The most glaring omission is estimates of the costs of reducing pollution. Empirical work overwhelmingly focuses on the benefits of reduction. For example, what decline in economic activity is necessary to achieve pollution reductions? As it stands, there are few results that would allow mayors to know whether policies are cost-effective or not.

More research is also needed for mayors outside of China and the US, especially in India – a large and populous country with relatively bad air quality. Institutional differences such as the ability and opportunity to migrate away from pollution or enforcement of policies may differ dramatically across countries and affect policy outcomes. As the work on driving restrictions has shown, historical antecedents such as the stock of used cars may affect outcomes and these antecedents will differ across countries.

Nonetheless, extant work has provided extensive evidence of a wide range of costs created by pollution going well beyond the traditional health and mortality costs that were historically examined. It has also identified numerous policies that are effective in reducing pollution: some forms of driving restrictions, appropriately-set congestion tolls, targeted public transit infrastructure, and incentive-aligned emissions standards. Mayors will also undoubtedly develop new policies such as promoting active commuting (walking or biking to work), subsidizing green vehicles, and water-canon trucks to suppress road dust[27] that will require evaluation.

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[27] For a description see "Chinese Cities are using this 'Mist Cannon' to Shoot Pollution from the Sky," *Quartz*, December 15, 2015 available at https://qz.com/572954/chinese-cities-are-using-this-mist-cannon-to-shoot-pollution-from-the-sky/ (accessed on April 14, 2021).

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