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ABSTRACT

Do Cities Mitigate or Exacerbate Environmental Damages to Health?*

Do environmental conditions pose greater health risks to individuals living in urban or rural areas? The answer is theoretically ambiguous: while urban areas have traditionally been associated with heightened exposure to environmental pollutants, the economies of scale and density inherent to urban environments offer unique opportunities for mitigating or adapting to these harmful exposures. To make progress on this question, we focus on the United States and consider how exposures—to air pollution, drinking water pollution, and extreme temperatures—and the response to those exposures differ across urban and rural settings. While prior studies have addressed some aspects of these issues, substantial gaps in knowledge remain, in large part due to historical deficiencies in monitoring and reporting, especially in rural areas. As a step toward closing these gaps, we present new evidence on urban-rural differences in air quality and population sensitivity to air pollution, leveraging recent advances in remote sensing measurement and machine learning. We find that the urban-rural gap in fine particulate matter (PM2.5) has converged over the last two decades and the remaining gap is small relative to the overall declines. Furthermore, we find that residents of urban counties are, on average, less vulnerable to the mortality effects of PM2.5 exposure. We also discuss promising areas for future research.

JEL Classification: I10, Q53, Q54
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1 Introduction

Urban areas—characterized by dense populations, concentrated industrial activity, and traffic—have traditionally been associated with heightened exposure to environmental pollutants. However, the economies of density and scale inherent to urban environments provide unique opportunities for mitigating and adapting to these harmful exposures, such as pollution reduction initiatives and improved access to health care facilities, potentially offsetting the adverse health effects of pollution. This dichotomy presents a complex and intriguing question: Do urban areas pose higher or lower environmental health risks compared to rural areas?

In this paper, we offer our perspective on the existing evidence and knowledge gaps concerning this question. Our discussion primarily focuses on the US, but the concepts are relevant to other settings as well, particularly in the developed world. Our starting insight is that the damages from poor environmental conditions hinge on two factors: exposure to pollutants and the shape of the damage function, which we refer to as vulnerability.

When comparing environmental health impacts in urban and rural areas, we focus on per capita effects rather than aggregate damages. This individual-level perspective provides insight into how a shift in population to urban areas might alter overall environmental damages. When considering environmental policies, it is essential to weigh both these per capita impacts and the population density of an area. Under economies of scale, investments in environmental improvements yield greater public health benefits in densely populated urban areas than in rural settings. Consequently, optimal policy would dictate a disproportionate allocation of these investments to urban areas, potentially reducing per capita environmental risk in cities relative to rural areas. Whether cities ultimately pose higher or lower environmental health risks compared to rural areas forms the central focus of our analysis and discussion.

In the first part of the paper, we examine evidence of disparities in exposure to adverse environmental conditions between urban and rural areas, primarily focusing on air quality, drinking water quality, and extreme temperatures. We review the existing literature on this topic and point out substantial gaps in knowledge, in large part due to historical monitoring and reporting deficiencies. As a step toward closing this gap, we present novel evidence on the gaps in pollution exposure between urban and rural areas, leveraging newly available remote sensing data. However, further research is needed in this area.

In the second part, we explore how population sensitivity to environmental conditions varies between urban and rural areas. We attribute this variation to differences in two broad categories: individual-specific factors, such as avoidance behavior or health stock, and place-specific factors, such as access to health care or public institutions that promote avoidance behaviors through the provision of information. We review existing studies, noting that most have not directly addressed urban-rural differences. To help fill this gap, we present new evidence on urban-rural gaps in population sensitivity to air pollution, using recent advances in machine learning techniques.

Historically, urban areas have likely faced greater health risks from environmental conditions than rural areas, including poorer air and drinking water quality. We spend the rest of the paper addressing how these dynamics have evolved in light of significant improvements in environmental conditions over recent decades. We conclude that it is still unclear whether cities mitigate or exacerbate the health damages caused by environmental factors.

2 Urban-Rural Differences in Exposure

In this section, we examine the existing evidence of disparities in urban and rural environmental conditions, focusing on air and drinking water pollution and ambient temperature, while briefly covering other factors. We also incorporate recent advancements in remote sensing techniques to present a novel analysis of urban-rural differences in air pollution levels and trends across the US from 1998 to 2020.

2.1 Air Pollution

Before the 1950s, air pollution in the US was largely unregulated. An escalation of severe air pollution episodes in urban areas in the mid-20th century illuminated the harms of air pollution and the need for monitoring and control measures. Los Angeles serves as a notable case. In the 1940s, a convergence of industrialization, population growth, and a rise in automobile use led to extreme air pollution and recurring episodes of smog so intense that visibility in the city was often reduced to just a few blocks (Haagen-Smit, 1970; Los Angeles Times, 1948). These circumstances prompted the establishment of the Los Angeles County Air Pollution Control District in 1947, the nation's first air pollution control program. Subsequent research and regulatory advancements culminated in 1970 with the passage of the Clean Air Act (CAA), which set national air quality standards, and the formation of the Environmental Protection Agency (EPA), which was given the authority to control air pollution (US Environmental Protection Agency, 2022a). The subsequent decades saw remarkable reductions in emissions and improvements in air quality. For instance, since the initiation of federal exhaust standards in 1967, emissions per mile from new vehicles in the US have plummeted by over 99% (Jacobsen et al., 2023), and the emissions of six key air pollutants—particulate matter, sulfur dioxide, nitrogen oxides, volatile organic compounds, carbon monoxide, and lead—have seen a combined 78% reduction since the CAA's passage (US Environmental Protection Agency, 2023).

Despite these advancements, findings from several studies indicate that air pollution levels generally remain higher in urban areas compared to rural ones. An analysis of environmental exposure data from the CDC's National Environmental Public Health Tracking Program for all counties in the contiguous United States from 2008 to 2012 found that air quality, as measured by violations of National Ambient Air Quality Standards (NAAQS) and average pollutant concentrations, is worse in more urban areas (Strosnider et al., 2017). The study found that ground-level concentrations of ozone and fine particulate matter ($PM_{2.5}$)—the air pollutants viewed by the EPA as posing the most widespread health threats (US Environmental Protection Agency, 2022b)—violated NAAQS standards in the most urban counties an average of 48 and 11 days per year, respectively, about 12 times greater than the number of violation days in the most rural counties. However, this study did not account for the more frequent pollution monitoring in urban counties, and thus comparing the raw number of violation days could overstate the actual difference in ambient air quality between urban and rural areas. Strosnider et al. (2017) also report that the mean annual $PM_{2.5}$ concentration was $11.2\,\mu g/m^3$ in the most urban counties, about $2.3\,\mu g/m^3$ (25%) higher than in the most rural counties. Several other studies focusing on specific regions of the US from 2000 to 2020 have also found mean concentrations of $PM_{2.5}$ and NO_2 that were higher in urban areas, with average $PM_{2.5}$ concentrations being higher in urban areas by amounts ranging from 3% to 50% (Berman and Ebisu, 2020; Clements et al., 2016; Garcia et al., 2016; Kundu and Stone, 2014).

Standard measures for assessing pollution concentrations may inaccurately portray disparities in harmful exposure between urban and rural areas, as they often overlook the differential compositions, or "speciation," of pollutants across areas. This is particularly true for $PM_{2.5}$, a critical air pollutant that the Institute for Health Metrics and Evaluation's Global Burden of Disease project has credited with causing over 90% of deaths due to outdoor air pollution (Health Effects Institute, 2020). $PM_{2.5}$ refers to airborne particles that are 2.5 micrometers or smaller in diameter and is measured and regulated based on its mass concentration in the atmosphere. However, $PM_{2.5}$ particles can include various components such as acids, organic chemicals, metals, soil or dust particles, and allergens, and some of them may be more harmful to health than others. As a result, when considering urban-rural disparities in $PM_{2.5}$ exposure, relying solely on mass concentrations may not sufficiently capture disparities in the potential harms.

For instance, Wang et al. (2022) analyzed $PM_{2.5}$ in the Midwest and focused on the cellular oxidative potential (OP), which measures the particle's ability to induce oxidative stress. The authors found that cellular OP levels were similar across rural, urban, and roadside sites, suggesting that the health risks associated with ambient $PM_{2.5}$ exposure may be similar in urban and rural areas despite higher mass concentrations in urban areas. In another study in Iowa from 2009 to 2012, researchers found that rural areas had more dirt and dust particles in the air, mostly from farming and unpaved roads, while city air had more pollution from vehicles and industrial activities (Kundu and Stone, 2014). These findings highlight the importance of additional research to understand the specific composition and characteristics of air pollutants across areas and to assess their potential health impacts.

While the improvement in air quality in cities is well-documented, it has been difficult to comprehensively compare air quality levels in urban versus rural areas due to the limited network of ground-based pollution monitors outside of cities. We address this gap using newly available spatially continuous satellite data on $PM_{2.5}$ levels for the period 1998–2020 from Van Donkelaar et al. (2021). We aggregate these high-resolution gridded data ($0.01° \times 0.01°$, approximately 1 km$\times$1 km) to zip-code tabulation areas (ZCTAs) in the US and then assign ZCTAs to one of five urban area groups using the US Census Bureau's Urban Area to ZCTA Relationship File.[1] The Census Bureau categorizes areas into three groups: "Rural" (not in an urban area), "Urban Clusters" (urban areas with a population between 2,000 and 50,000), and "Urbanized Areas" (urban areas with a population over 50,000). We use these standard categories but split the Urbanized Areas category into three based on population size (50K–1M, 1M–5M, and over 5M) so that we can differentiate between small, medium, and large cities. The largest category (over 5 million) represents the six largest urban areas in the US: Chicago (IL, IN), Dallas–Fort Worth–Arlington (TX), Los Angeles–Long Beach–Anaheim (CA), Miami (FL), New York–Newark (NY, NJ, CT), and Philadelphia (PA, NJ, DE, MD).

Figure 1 plots average $PM_{2.5}$ levels (µg/m^3) over time for these five urban categories, revealing several key findings. First, there is a clear gradient of air pollution between the most rural and most urban areas, with larger urban areas consistently exhibiting higher levels of $PM_{2.5}$. Second, the well-known decline in $PM_{2.5}$ over the past two decades is apparent across all levels of urbanicity. Even in rural areas, average $PM_{2.5}$ levels have been nearly cut in half since 1998. Third, the gap in average $PM_{2.5}$ levels between the most urban and most rural areas is small relative to the overall trends. For example, the average $PM_{2.5}$ level in the most urban areas in the last five years of the sample (8.57 µg/m^3 in 2016–2020) was 23% lower than the average $PM_{2.5}$ level in rural areas in the first five years (11.07 µg/m^3 in 1998–2002). Fourth, there has been some convergence between urban and rural areas since 1998. The gap between the most urban areas and rural areas declined by 42% between the first five years (3.58 µg/m^3 in 1998–2002) and the last five years of the sample (2.09 µg/m^3 in 2016–2020). This convergence is likely due in part to the 2005 implementation of NAAQS for $PM_{2.5}$, which Currie et al. (2023) show led to substantial declines in average $PM_{2.5}$ levels for non-attainment counties (event studies show long-term declines up to 2 µg/m^3 by 2012).

[1] ZCTA boundaries do not overlap perfectly with urban area boundaries. For classification purposes, ZCTAs are assigned to the most urban group in which any of their population resides. As such, a ZCTA is categorized as "Rural" (the least urban group) only if none of its population resides in one of the more urban groups. The relationship file is available here: https://www.census.gov/geographies/reference-files/2010/geo/relationship-files.html.

How do we expect differences in urban and rural air pollution levels to evolve going forward? Given current trends and regulatory developments, there are at least three reasons to expect further convergence. First, in January 2023, the EPA announced plans to revise its annual $PM_{2.5}$ NAAQS from its current level of 12 μg/m^3 to a new level between 9 and 10 μg/m^3.[2] This new rule will be binding primarily for a set of high-pollution cities. If the 2005 NAAQS implementation is any indication, it is likely to bring further convergence between urban and rural areas. Second, one of the major sources of urban air pollution—vehicle exhaust—is likely to significantly diminish as consumers shift to electric vehicles in the coming years. Indeed, much of the urban air pollution generated by internal combustion vehicles will likely shift to more rural areas, where electric power is generated (Holland et al., 2016). This may be largely avoided if a shift away from highly polluting electric power generation (e.g., coal) keeps up with the shift toward electric vehicles. Finally, pollution from wildfire smoke is rapidly becoming a dominant source of air pollution (Burke et al., 2021), and recent evidence shows that rural county residents experience more days with wildfire smoke exposure compared to their urban counterparts (Molitor et al., 2023).

2.2 Drinking Water Pollution

The history of drinking water pollution in the US dates back to the late 19th century, when waterborne diseases like cholera and typhoid spread rapidly due to contaminated water sources, prompting the need for improved sanitation (Anderson et al., 2021; Cutler and Miller, 2005). However, it was not until the mid-20th century that nationwide drinking water regulations were established. In the 1960s and 1970s, notable events such as the contamination of the Hudson River by industrial pollutants and the Love Canal incident, where a neighborhood was built over a toxic waste dump, contributed to increased public awareness of water pollution. Pressure to protect drinking water from toxic contaminants ultimately resulted in the 1974 passage of the Safe Drinking Water Act (SDWA), which authorized the EPA to set national standards for drinking water to protect against both naturally occurring and man-made pollutants. Since then, numerous amendments to the SDWA have been made, each in response to new challenges such as the discovery of new

[2] See https://www.epa.gov/pm-pollution/national-ambient-air-quality-standards-naaqs-pm.

contaminants, further highlighting the ongoing importance of drinking water regulation.

Despite the SDWA's enactment and its subsequent amendments, our understanding of the prevalence and trends in drinking water contamination across the US remains limited. Unlike the progress made in air pollution monitoring after the CAA's passage, which resulted in a wealth of data on air pollutant concentrations and the widespread recognition of their adverse health effects, the impacts of the SDWA on drinking water quality trends, as well as the resulting impacts on health outcomes, are less well-documented and understood.

Part of this knowledge gap on drinking water contamination stems from a lack of data. Prior studies have examined data on drinking water violations that occur when concentrations of a regulated contaminant exceed the maximum contaminant level defined by the EPA. One study across 26 states during 2010–2015 found that drinking water violation rates were lower in urban areas than in rural areas: across 10 regulated contaminants, approximately 5.4% of systems in large metropolitan areas and up to 10% in the most rural areas reported at least one violation (Strosnider et al., 2017). Another nationwide study of community water systems between 1982 and 2015 found that the rate of all violation types in rural counties is 76% higher than in urban counties (Allaire et al., 2018). These findings suggest that drinking water pollution is generally lower in urban areas, offering a stark contrast to the pattern of greater air pollution in urban areas noted above and highlighting one area in which environmental investments have led urban areas to become cleaner than their rural counterparts.

Despite the insights gained from studies on SDWA violations, a significant knowledge gap remains in our understanding of the overall levels, trends, and urban-rural disparities in drinking water quality. This is largely due to the lack of systematic reporting on contamination levels in drinking water. Violation rates only indicate if standards are met or not, without capturing changes in pollution levels that stay above or below those standards. Violation reports also provide no information on unregulated pollutants. Changes in violation rates can indicate changes in pollution levels, regulatory standards, or monitoring and reporting behavior. Private wells, which serve 11% of US households with a greater prevalence in rural (42%) than urban areas (3%) (US Census Bureau, 2021), are exempt from federal and most state monitoring and reporting requirements. A nationwide assessment found that

23% of these wells exceeded contamination benchmarks, indicating that the rural-urban water quality gap may be larger than suggested by violation reports or other regulatory data (DeSimone et al., 2009).

There is a pressing need for more comprehensive data on drinking water quality as well as research into both the health impacts of water contamination and the role of legislative measures like the SDWA, and other factors, in contributing to disparities in drinking water contamination. Addressing this need, a recent study collected data on the concentration of pollutants in drinking water of public water systems across the United States. This study linked the data to the characteristics and health outcomes of the populations served by these systems, offering a granular view of the trends, causes, and impacts of water pollution in the US (Keiser et al., 2023). The study revealed that larger systems, serving major metro areas like Los Angeles and Chicago, typically have lower pollution levels than those serving smaller populations. This trend toward cleaner drinking water in cities may be influenced by stricter SDWA standards for larger systems and the economies of scale in pollution abatement.

2.3 Temperature

Temperature is another environmental stressor known to differ between urban and rural areas. Urban areas typically experience higher surface and air temperatures than their rural counterparts due to a phenomenon known as the urban heat island (UHI) effect. This effect arises from various factors such as reduced evaporative cooling due to less vegetation, decreased solar radiation reflection, greater heat retention in buildings and roads, anthropogenic heat emissions, and alterations in energy convection to the lower atmosphere (Zhao et al., 2014). While the predominant causes of UHI remain subject to ongoing discussion, the existence of this phenomenon is firmly established.

UHIs are usually quantified as the temperature difference between an urban area and a nearby non-urban area. Although there is no consensus for a standardized way of measuring UHIs (Stewart, 2011), they can be measured in terms of air temperatures (typically using ground-based weather stations) or remotely sensed surface temperatures (Schwarz et al., 2012). Remote sensing offers an advantage in that UHI measurements can be spatially continuous and highly granular. As such, many recent studies of UHIs rely on remotely sensed data and measure surface UHI effects. That being said, air temperatures are more relevant for studying the health implications of UHIs.

UHI effects have been extensively studied. Studies range in scope from local studies focusing on individual cities to national (Chakraborty et al., 2020; Imhoff et al., 2010) and global investigations (Peng et al., 2012). A key finding from this literature is that UHI effects can significantly vary across cities. For example, in a global analysis of 419 cities, Peng et al. (2012) found an average daytime surface UHI effect of about 1.5°C, with the largest effect of 7.0°C seen in Medellín, Colombia. Another analysis of 38 US cities revealed an average daytime surface UHI of 2.9°C (Imhoff et al., 2010).

UHI effects exhibit diurnal and seasonal fluctuations, but the extent and direction of these fluctuations depend on the specific setting and study methods. Despite the variable magnitudes, studies consistently report evidence of UHI effects during winter and summer seasons as well as day and night. The timing of UHI effects is of central importance to their health implications.

While numerous studies estimate the health impacts of UHIs, the broad implications are far from clear. Many studies focusing on summer mortality find that UHIs lead to serious health consequences. For example, Iungman et al. (2023) find that they are responsible for over 4% of all summer deaths in European cities. A smaller number of studies have considered potential protective effects of UHIs in the winter. For example, Macintyre et al. (2021) focus on a specific region in the UK and find that UHIs may prevent more cold-related deaths than they cause heat-related deaths. Both these and many other studies of UHIs infer the health impacts by combining UHI intensity estimates with pre-existing dose-response functions. However, this approach assumes a fixed dose-response relationship across areas and thus may either over- or understate the disparate impacts of heat in urban versus rural areas. A valuable direction for future research is to estimate the health effects of UHIs directly, using dose-response relationships specific to urban and rural areas. Given the lack of consistent evidence and limited economic studies on UHIs, we see an opportunity for applied microeconomists to contribute to this area of research.

2.4 Other Environmental Conditions

Recent studies suggest that other environmental stressors such as noise and light pollution can also significantly and negatively impact health, particularly mental health, possibly due at least in part to sleep disruption (Boslett et al., 2021; Hener, 2022; Jones, 2018; Zou, 2017). Urban areas, with their generally higher levels of noise and light, may pose greater health risks of this nature compared to rural settings. However, rural areas are not necessarily immune to the harms of these types of pollutants. For example, wind farms, commonly sited in rural areas, generate a large amount of low-frequency noise that has been linked to increased suicide rates (Zou, 2017). More research is needed to determine which types of noise and light pollution are the most harmful and how these exposures differ across rural and urban environments.

3 Urban-Rural Differences in Vulnerability

Section 2 focused on differences in exposure to poor environmental conditions between urban and rural populations. Total damages from poor environmental conditions depend on both exposure and the shape of the damage function. For example, suppose that the damage function for air pollution is steeper for rural populations (i.e., the marginal damage from another unit of air pollution is larger). If this is the case, then the total damages from air pollution may be larger for rural populations despite their lower level of exposure.

Why could damage functions differ across urban and rural populations? We consider two categories: place-specific factors (e.g., health care access) and individual-specific factors (e.g., health stock). We think of place-specific factors as causal determinants of marginal damages. If a rural resident moved to an urban location, we would expect their vulnerability to environmental conditions to change due to place-specific factors. Individual-specific factors, on the other hand, represent selection across urban and rural locations. In what follows, we consider urban-rural differences in individual- and place-specific factors in turn.

3.1 Individual-Specific Factors

We begin our discussion of individual-specific factors with an empirical exercise that builds on Deryugina et al. (2019, 2021). Deryugina et al. (2019) use detailed data from Medicare in the US to build a machine learning model to predict individual-level vulnerability to dying from acute exposure to $PM_{2.5}$, documenting geographic heterogeneity in their measure of vulnerability. We use data from Deryugina et al. (2021) at the ZCTA level to document differences in $PM_{2.5}$ vulnerability across urban and rural locations. Figure 2 presents the results, showing two outcomes: the share of individuals "vulnerable" to $PM_{2.5}$ (in the top 25% of the predicted vulnerability distribution) in Figure 2A, and the share of individuals "extremely vulnerable" (in the top 1% of the predicted vulnerability distribution) in Figure 2B. For both outcomes, vulnerability tends to be larger in more rural ZCTAs. For example, the share of extremely vulnerable Medicare beneficiaries is 37% larger in rural ZCTAs compared to the most urban ZCTAs.

This analysis shows that in the Medicare population, the pollution-related health stock among rural residents is lower compared to their more urban counterparts. While our analysis is specific to mortality from $PM_{2.5}$ exposure, a large literature on urban-rural differences in health suggests that urban populations likely have better health stock on several dimensions. For example, using data from 1999 to 2021 in the US, Curtin and Spencer (2021) document lower mortality rates among urban counties for each of the 10 leading causes of death. A better health stock along a range of dimensions would make urban populations more resilient to the impacts of different types of environmental shocks (e.g., extreme heat) on a range of different health outcomes (e.g., mental health).

Until this point, we have used the term "exposure" in reference to ambient environmental conditions. However, the same ambient conditions can translate into very different exposures at the individual level. For example, Burke et al. (2022) find that indoor $PM_{2.5}$ concentrations can vary by a factor of 20 for neighboring households exposed to the same wildfire smoke event. Differences in individual-level exposure to environmental conditions depend on defensive behavior.[3] To our knowledge, there is little direct evidence of urban-

[3] We use "defensive behavior" as a catch-all term for defensive investments, avoidance behavior, and adaptive responses.

rural differences in defensive behavior. However, there are several reasons to suspect that rural individuals are less likely to engage in defensive behavior. Income is a key factor here: rural populations have lower average incomes compared to their urban counterparts, and a building body of evidence suggests that lower-income populations are less likely to engage in defensive behavior against a range of environmental shocks.

Access to air conditioning can dramatically mitigate the health consequences of extreme heat (Barreca et al., 2016), yet energy expenditures (due to using air conditioning) of low-income households are much less responsive to heat events (Doremus et al., 2022). Furthermore, using global data, Carleton et al. (2022) find that the heat-mortality relationship is much stronger in lower-income countries. Similar results emerge in the literature on defensive behavior in response to other environmental shocks. Burke et al. (2022) find that during wildfire smoke events, residents of lower-income counties are less likely to search for information about air quality and are less likely to stay home compared to those of higher-income counties. Marcus (2022) finds that following public notice of water quality violations, lower-income households are less likely to purchase bottled water relative to higher-income households.

Income is not the only channel likely to generate differences in individual-level vulnerability across urban and rural populations. Occupational exposure to poor environmental conditions is likely to be higher among rural populations, who spend significantly more time outdoors and are more likely to be employed in outdoor occupations (Matz et al., 2015). Emerging evidence shows a link between environmental conditions and occupational health claims that is particularly strong among outdoor industries and occupations (Dillender, 2021; Ireland et al., 2023; Park et al., 2021). Finally, attitudes about environmental risk may play an important role. If rural residents perceive a lower risk from being exposed to poor environmental conditions, then they will be less likely to take protective behavior. Prior work has found that Republicans (a political affiliation correlated with rural geography in the US) are less likely to make protective investments against climate-related risks (Botzen et al., 2016).

3.2 Place-Specific Factors

The prior subsection spoke to whether the types of people who live in cities are more or less vulnerable to environmental shocks. This subsection considers whether cities and the amenities they offer can affect vulnerability. There is reason to believe this could be the case. For example, Kahn (2005) finds that countries with higher-quality institutions tend to suffer fewer deaths following a natural disaster. While Kahn (2005) is a cross-country analysis, the same logic may apply to the setting in this paper if cities tend to have better institutions relevant to the health impacts of environmental shocks.

It is plausible that local institutions could affect environmental vulnerability through at least two channels. They could facilitate defensive behaviors, thereby reducing individual-level exposure to environmental shocks, or conditional on exposure, they could provide services such as improved health care to mitigate damages from exposure.

Can cities facilitate defensive behavior? Information provision is an area in which cities may have an advantage. Air quality warning systems have been shown to increase defensive behaviors (Neidell, 2009; Saberian et al., 2017; Zivin and Neidell, 2009), and these systems are typically managed by local authorities that are likely better funded in more populous areas. A similar argument could also be made for heat (and cold) warning systems. While the US National Weather Service provides consistent heat warnings across the country based on temperature forecasts, research has criticized these traditional meteorological-based warning systems (Li et al., 2022). As such, better-equipped local authorities may offer higher-quality warning systems. While it is logical that there could be economies of scale in the provision of information, we emphasize that this is speculative: we are unaware of any research that explicitly documents variation in information about environmental conditions across rural and urban areas.

Population density in cities may also offer an energy cost advantage. Ross et al. (2018) find that rural households spend a larger share of their income on energy bills, reflecting differences between rural and urban settings in housing and other consumption choices as well as in available energy types and their prices. The most evident difference is in heating costs, as many rural homes lack access to natural gas and rely on alternatives like heating oil or propane. Whether urban households enjoy an energy price advantage depends on the price differences between natural gas and these alternatives. In recent years, there has been a substantial energy price advantage associated with natural gas (Myers, 2019).

Substantial evidence suggests that cities have an advantage in providing health care. Residents of urban areas, on average, live closer to hospitals (Lam et al., 2018). Proximity to care is especially important for time-sensitive conditions such as stroke and heart attack (Carroll, 2019; Gujral and Basu, 2019), which represent a large share of deaths attributable to poor environmental conditions (Cohen et al., 2017). Urban residents also have access to higher-quality care (Dingel et al., 2023; Fischer et al., 2022; Joynt et al., 2011). To what degree is improved access important for mitigating health damages from environmental shocks? A small but growing literature suggests it is important. Liao et al. (2023) and Sarmiento (2023) find that the effects of temperature on mortality are smaller in areas with access to higher-quality health care in China and Colombia, respectively. Mullins and White (2020) take a causal approach to this question, exploiting quasi-experimental variation in access to primary care using the staggered rollout of US community health centers in the 1960s. They find that access to community health centers—rolled out primarily in urban counties—moderated the heat-mortality relationship by about 15%.

As a final point, we acknowledge there are important interactions between individual- and place-specific factors. If an individual moves to a city, we expect their vulnerability to environmental conditions could change immediately due to the factors already discussed in this subsection. However, in the long term, living in a city can have a causal effect on individual-specific factors such as health stock (e.g., by offering higher-quality health care) or risk perceptions (through information or peer effects).

4 Conclusion

From an environmental perspective, is it worse for your health to live in a city? In our view, the answer is far from clear. Robust evidence suggests that urbanites are exposed to more air pollution, though we find that this gap has shrunk over the past two decades and it is reasonable to expect it to shrink further going forward. Given the increasing adoption of electric vehicles, there is potential for future researchers to examine how much this transition will shift air pollution away from highly populated urban centers. Current evidence suggests that urban residents are less exposed to poor water quality, though this research has been hampered by a lack of comprehensive data. For urban heat, most of the current research points in the direction of negative health impacts for urban residents, but there is a lack of large-scale research directly examining these health consequences. Given that the mortality effects of heat have reduced dramatically in recent decades while the mortality effects of cold have not (Deschenes, 2022), it is plausible that the protective effects of winter urban heat offset or even outweigh the harmful effects of summer urban heat. The health impacts of urban heat is an area that seems ripe for further research.

The evidence on urban-rural gaps in vulnerability is currently more one-sided, suggesting that urban residents are less vulnerable to environmental shocks. While this is in part due to selection—urban residents are higher income and healthier, on average—there is also evidence that the amenities provided in an urban setting, such as closer proximity to health care, can mitigate the impacts of poor environmental conditions. While there is a small literature on the protective effects of access to, and quality of, health care, future work could examine whether other local amenities have a protective effect as well.

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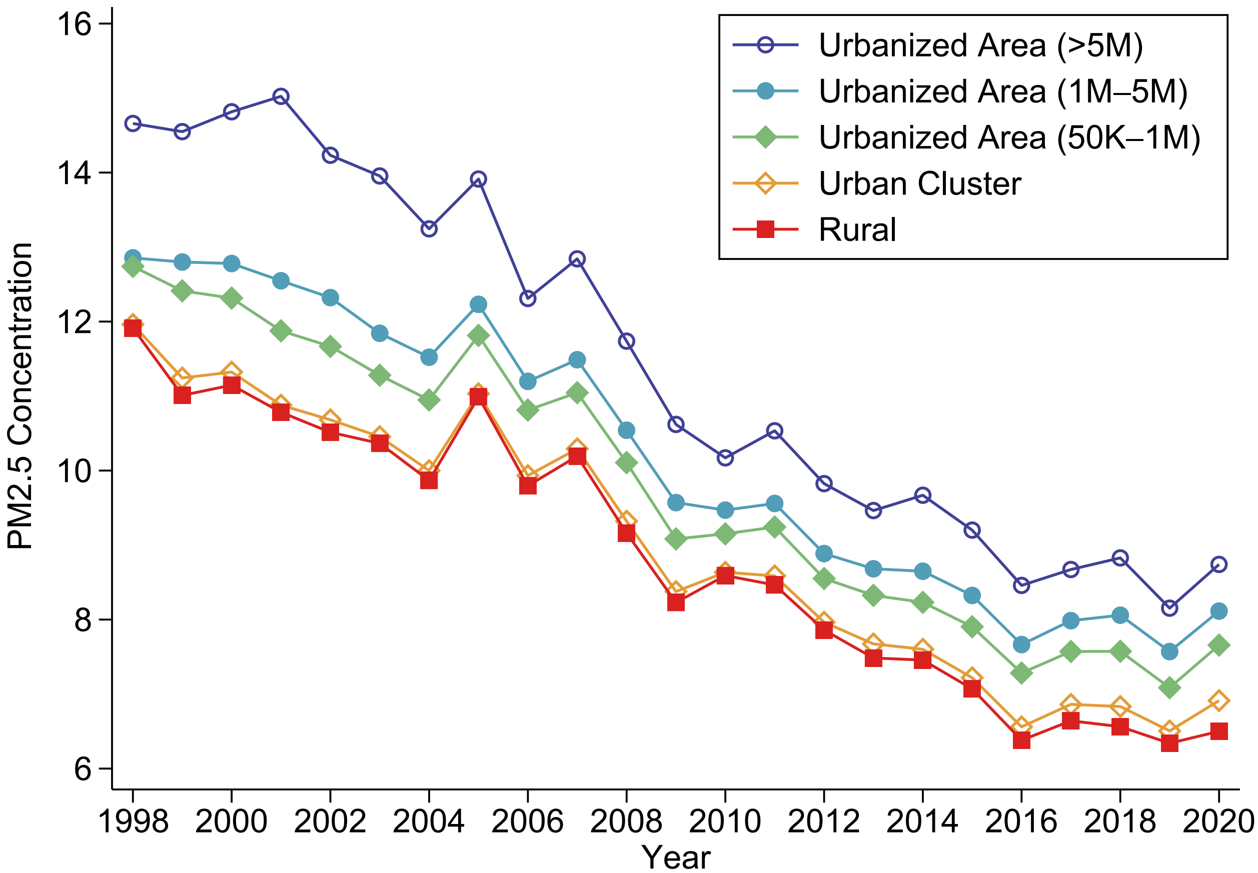
Figures

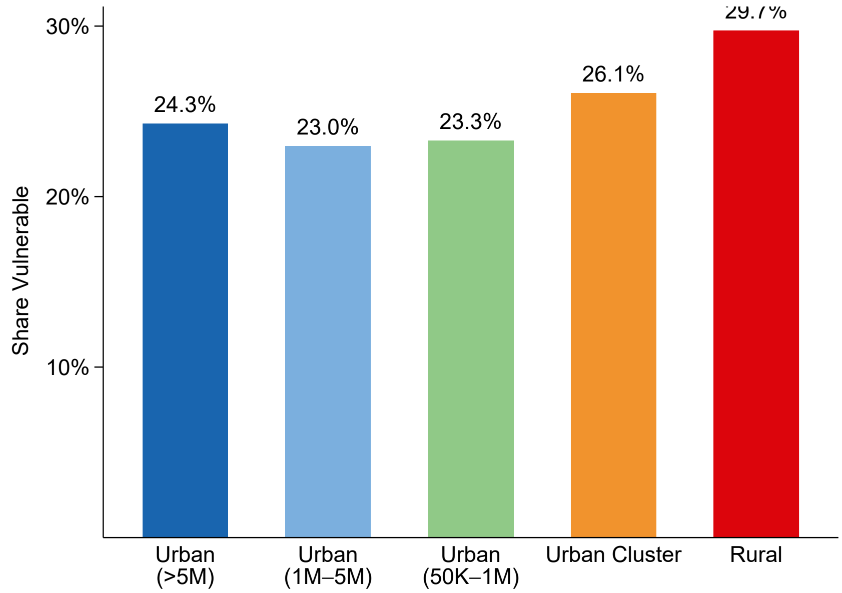
Figure 1: Average $PM_{2.5}$ Concentration by Urbanicity in the US, 1998–2020



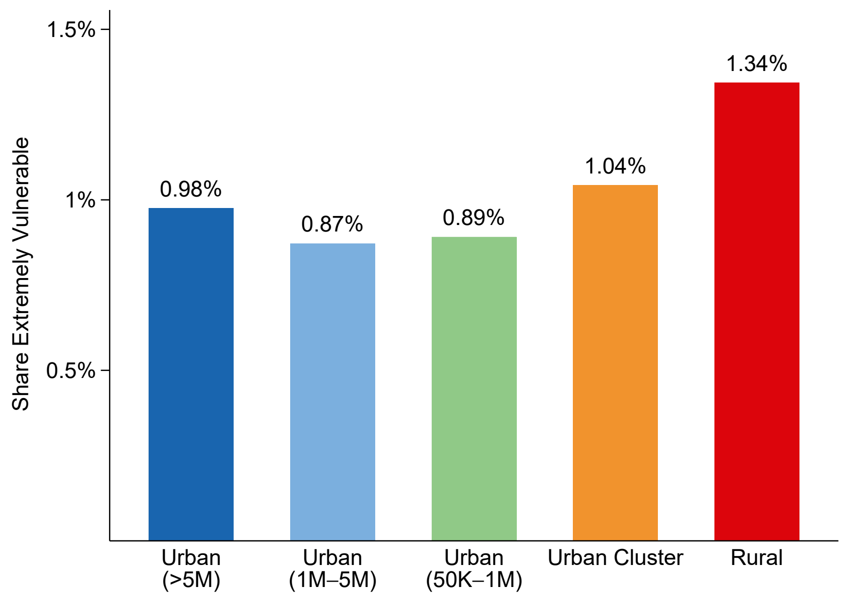
Notes: Spatially continuous gridded (0.01° × 0.01°) data on $PM_{2.5}$ are derived from Van Donkelaar et al. (2021). We aggregate these data to the ZCTA level and then assign ZCTAs to one of five groups based on the US Census Bureau's definition of urban areas. The Census Bureau classifies places into three groups: Rural (not in an urban area), Urban Clusters (urban areas with a population of 2,000–50,000), and Urbanized Areas (urban areas with a population over 50,000). We split the Urbanized Area category into three groups based on population size (50K–1M, 1M–5M, and over 5M) to differentiate between small, medium, and large cities.

Figure 2: Predicted Vulnerability to $PM_{2.5}$ by Urbanicity in the US

(A) Vulnerable (Top 25%)



(B) Extremely Vulnerable (Top 1%)



Notes: This figure reports predicted vulnerability to $PM_{2.5}$ exposure by urbanicity in the US. The data are derived from Deryugina et al. (2019), who combine detailed Medicare data on US adults aged 65 and older with machine learning methods to predict individual-level vulnerability to mortality from acute exposure to $PM_{2.5}$. These data were aggregated to the ZCTA level by Deryugina et al. (2021) to analyze geographic heterogeneity in vulnerability. We further processed these ZCTA-level data to calculate vulnerability by urbanicity, assigning ZCTAs to one of five urban groups following the same procedure and group definitions as in Figure 1.