ENVIRONMENTAL STUDIES

Air pollution and visitation at U.S. national parks

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Hundreds of millions of visitors travel to U.S. national parks every year to visit America's iconic landscapes. Concerns about air quality in these areas have led to strict, yet controversial pollution control policies. We document pollution trends in U.S. national parks and estimate the relationship between pollution and park visitation. From 1990 to 2014, average ozone concentrations in national parks were statistically indistinguishable from the 20 largest U.S. metropolitan areas. Further, relative to U.S. cities, national parks have seen only modest reductions in days with ozone concentrations exceeding levels deemed unhealthy by the U.S. Environmental Protection Agency. We find a robust, negative relationship between in-park ozone concentrations and park visitation. Still, 35% of all national park visits occur when ozone levels are elevated.

INTRODUCTION

The year 2016 marked the 100th year anniversary of the U.S. National Park Service (NPS) (*1*). Created by the Organic Act of 1916, the NPS serves more than 300 million visitors annually across all of its locations. A recent survey found that nearly 90% of respondents had visited a national park area in their lifetime, and a third of respondents anticipated visiting a park in the coming year (*2*).

The Organic Act established the NPS mission to "leave [parks] unimpaired for the enjoyment of future generations" (*1*, *3*). Several federal legislative efforts have sought to address environmental quality in national parks. Prominent efforts include the Clean Air Act (CAA) Amendments of 1977 and 1990, which protect parks through a special designation as Federal Class I areas, and the Environmental Protection Agency's (EPA) Regional Haze Rule, which seeks to improve air quality and visibility in Class I areas (*4*–*6*). Ozone, the focus of this study, has been linked to numerous adverse human health outcomes, particularly during physical activity (*7*). Ozone is also correlated with decreased visibility (*8*) and contributes to vegetative damage (*3*, *9*, *10*). Recently, the NPS and advocacy groups have drawn attention to the high levels of ozone and poor visibility at national parks (*11*–*13*). Despite the known damages from ozone pollution, recent rules and regulations to control regional haze and ozone have been costly and controversial (*14*–*17*).

Here, we compile an extensive data set on visitation and air pollution at national parks to inform these policy debates. We focus on ozone, as it is the most widely monitored pollutant in national parks, and it is used to notify visitors of air quality conditions in parks. We address two primary research questions. First, we ask how ozone levels have changed at national parks, and how they compare to major U.S. metropolitan areas. Previous research has documented air quality and ozone trends for individual parks, urban versus rural areas, and the entire United States (*18*–*23*). We present annual trends in both ozone concentrations and ozone levels exceeding those deemed unhealthy by the EPA. Our methods explicitly control for differences in local weather conditions and seasonality to isolate changes in ozone concentrations due to anthropogenic sources. We compare these trends to those in major U.S. metropolitan areas. Second, we estimate the association between ozone and visitation at national parks and explore potential mechanisms to understand the implications of air quality improvements.

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RESULTS

Park and city ozone trends

Figure 1 presents average annual and summertime ozone trends for the 33 national parks in our sample, as well as for the 20 largest U.S. metropolitan areas from 1990 to 2014. Parks included in our sample provide broad coverage across the United States and include the largest and most heavily visited parks in the NPS system such as Acadia, Great Smoky Mountains, Yellowstone, and Yosemite. A full list is available in table S1. We estimate annual trends using daily pollution monitor readings and control for seasonal fluctuations, daily weather conditions, and location-specific characteristics. We use two measures of ozone pollution. First, we estimate annual trends in the maximum daily 8-hour ozone concentrations in each location. Maximum daily 8-hour ozone concentrations are used by EPA to establish ambient air quality standards under the CAA. The second measure is exceedance days, defined as the number of days in a year where the maximum daily 8-hour ozone concentration exceeds 70 parts per billion (ppb), a level deemed as "unhealthy for sensitive groups" by the EPA (*24*). Sensitive groups include children, older adults, and people with lung disease.

Average annual ozone concentrations in national parks are statistically indistinguishable from those in metropolitan areas for most of our sample. In 1990, metropolitan areas had higher average ozone concentrations, particularly in the summertime, and the average number of exceedance days in cities far exceeded those in national parks (Fig. 1, A to C). Summertime ozone concentrations and the average number of unhealthy ozone days are nearly identical in national parks and metropolitan areas starting in the 2000s. Average summer ozone concentrations decreased by more than 13% from 1990 to 2014 in metropolitan areas. Meanwhile, summertime ozone levels increased in parks from 1990 to the early 2000s and decreased thereafter to 1990 levels by 2014 (Fig. 1B). Over this same period, the average number of exceedance days in metropolitan areas fell from 53 to 18 days per year. National parks saw less progress, where average exceedance days decreased from 27 to 16 days per year. Ozone pollution at the national park with the highest average ozone concentrations, Sequoia National Park, follows a similar trend in exceedance days as the metropolitan area with the highest ozone concentrations, Los Angeles (Fig. 1D). Notably, exceedance days at Sequoia have surpassed those in Los Angeles in all but 2 years since 1996.

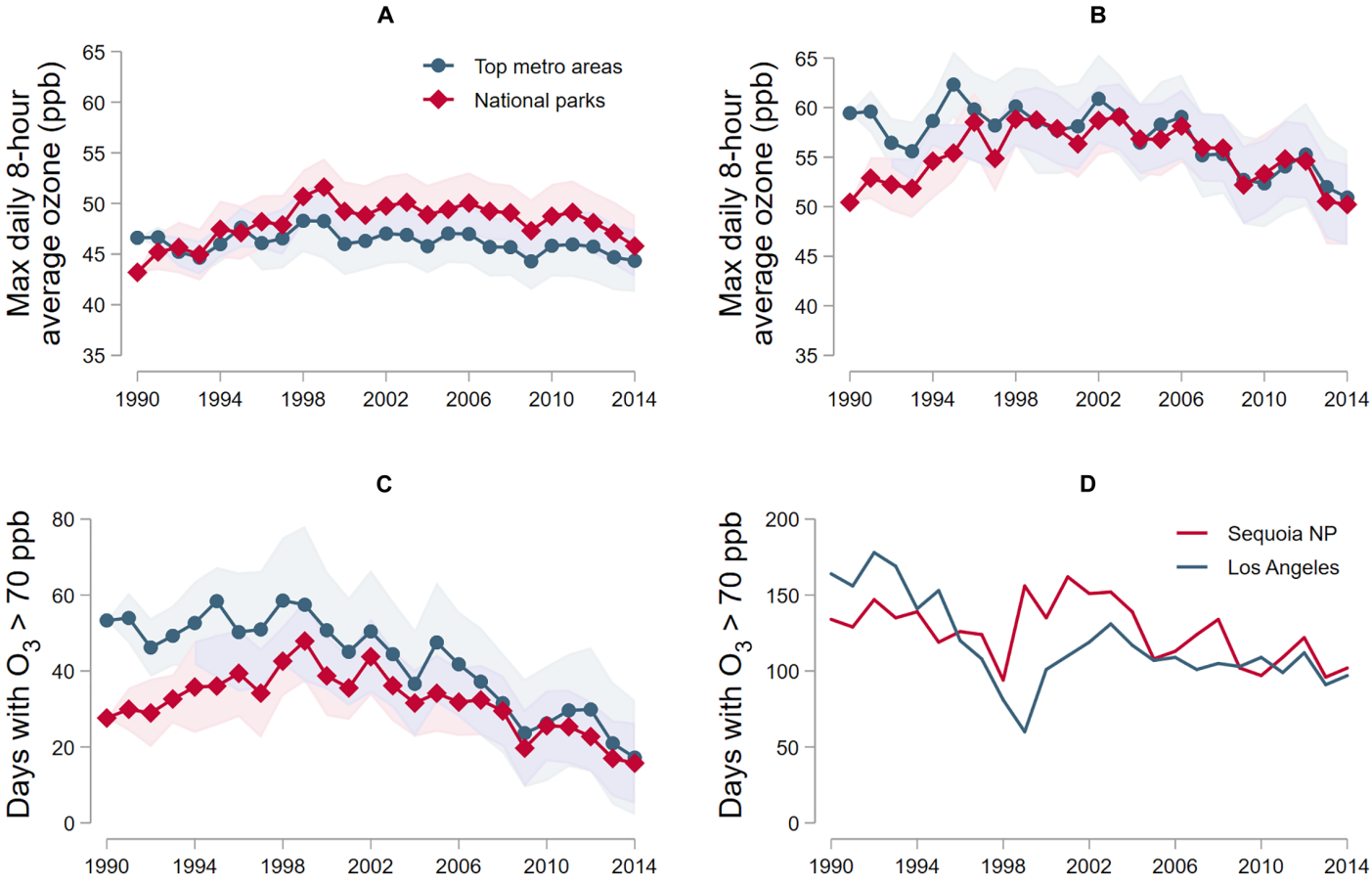

Fig. 1. Trends in maximum daily 8-hour ozone concentrations and days with maximum daily 8-hour ozone exceeding 70 ppb in large metro areas and national parks. (**A**) Average annual maximum daily 8-hour ozone concentrations. (**B**) Average summertime maximum daily 8-hour ozone concentrations. (**C**) Average days per year with maximum daily 8-hour ozone concentrations exceeding 70 ppb. (**D**) Exceedance days at Sequoia National Park and the Los Angeles metro area. (A to C) Control for seasonality, daily weather conditions, and park- and metro-specific fixed unobservable characteristics. Shaded areas are 95% confidence intervals.

Ozone and visitation at national parks

To examine how air pollution affects park visitation, we combine monthly park-level visitation statistics with monitor readings of ozone pollution at parks and use fixed-effect and instrumental variable (IV) estimation strategies. We use the monthly average of daily maximum 8-hour ozone concentrations in each park (maximum ozone) as our ozone pollution measure. Alternative specifications are shown in the Supplementary Materials. All results use data from 1990 to 2014 for the same 33 parks included in our trend analysis.

Column 1 of Table 1 presents the fixed-effects estimate of the response of monthly log visitation to maximum ozone. All regressions in Table 1 control for weather, common seasonal factors across parks (month fixed effects), and unobserved annual factors specific to each park (park-by-year fixed effects). The fixed-effect (FE) estimate suggests that a 1-ppb increase in maximum ozone is associated with a 1.6% decrease in monthly park visitation.

Automobiles are a significant source of nitrous oxides (NO_x), and both cars and vegetation produce volatile organic compounds (VOCs). Both are chemical precursors to ozone (*4*, *25*). Automobiles at national parks emit these precursors, potentially biasing our estimates of visitation responses. Specifically, if visitation causes more ozone in parks, our FE estimates will be biased downward in magnitude, and we will understate the negative association between pollution and visitation. We use an IV strategy to address this confounding effect and to address potential measurement error in park ozone readings. We use the average of maximum ozone in 20 upwind counties from each national park as our instrument for monthly park ozone concentrations. Column 2 of Table 1 presents the IV estimates. A 1-ppb increase in maximum ozone is associated with a 3.9% decrease in monthly park visitation. The larger estimated impact from the IV specification is consistent with visitation contributing to ozone formation in parks and measurement error.

Column 3 of Table 1 and Fig. 2A display estimates from the same regression as in column 1 but allow maximum ozone to have a differential effect across seasons. We find the largest negative estimates in the summer and fall, where a 1-ppb increase in maximum ozone is associated with a 2 and 1.5% reduction in visitation, respectively. Also, park visitation is highest during these two seasons. The seasonal effects are not estimated using the IV strategy and, therefore, are likely lower-bound effects.

Mechanisms for a negative ozone-visitation association

We explore two potential mechanisms to explain the negative association between ozone and park visitation. First, visitors may respond to Air Quality Index (AQI) warnings. Many national parks issue AQI warnings when ozone exceeds certain levels. A significant body of

Table 1. Estimated impact of monthly average maximum ozone concentrations (parts per billion) in national parks on log visitation from 1990 to 2014. All specifications include weather controls, park-by-year FE and month-of-year FEs. Column 2 instruments in-park monthly average maximum ozone using ozone concentrations from upwind counties. Column 3 examines the effects of ozone by season. Values in parentheses are robust SEs (standard errors) clustered at the park using a bootstrap procedure with 500 replications. ***$P < 0.01$, **$P < 0.05$, and *$P < 0.1$. The Stock-Yogo weak identification critical value for the Kleibergen-Paap F test is 16.38 for a 10% maximal IV bias relative to ordinary least squares.

Maximum ozone (ppb)	−0.0162**	−0.0394**	
	(0.00744)	(0.0165)	
Summer maximum ozone (ppb)			−0.0198**
			(0.00887)
Fall maximum ozone (ppb)			−0.0149**
			(0.00599)
Spring maximum ozone (ppb)			0.00198
			(0.00635)
Winter maximum ozone (ppb)			0.0194
			(0.0165)
Observations (N)	5603	5603	5603
Number of parks	33	33	33
Average maximum ozone (ppb)	47.51	47.51	47.51
Weather controls	Yes	Yes	Yes
Month FE	Yes	Yes	Yes
Park-by-year FE	Yes	Yes	Yes
IV	No	Yes	No
Kleibergen-Paap F test	—	95.85	—
R^2	0.904	0.900	0.908

research suggests that air quality warnings cause pollution avoidance behavior (*26*–*28*). A second potential mechanism is that visitors decrease visitation on days with poor visibility. The same pollutants that contribute to ground-level ozone can also reduce visibility (*29*). These two mechanisms correspond to targeted outcomes of the CAA and the Regional Haze Rule. The CAA places greater emphasis on reducing the incidence of exceedance days, whereas the Regional Haze Rule seeks to improve visibility. To explore these mechanisms, we estimate the response of monthly visitation at national parks to three variables: (i) the percentage of exceedance days in a month (exceedance day fraction), (ii) the average monthly visibility measured by park visibility monitors, and (iii) the average maximum visibility.

Column 1 of Table 2 presents results from our FEs estimation of the relationship between the logarithm of monthly visitation and fraction of ozone exceedance days in each month. An additional three exceedance days in a month is associated with an 8% decrease in visitation. Column 2 presents results from our IV specification using ozone concentrations in upwind counties. We estimate that an additional three exceedance days is associated with a 27% decrease in monthly visitation.

Figure 2 (B and C) provides additional evidence supporting the AQI warning mechanism. Figure 2B shows histograms of maximum daily 8-hour average ozone levels by season. Red vertical lines indicate the 70-ppb exceedance day threshold. The vast majority of winter readings are below the 70-ppb threshold. In spring, the distribution shifts right, although few days ever exceed 70 ppb. In contrast, the summertime distribution flattens, yielding more observations in the 70- to 100-ppb range. In fall, the distribution shifts back to the left. While average ozone concentrations are lower in fall, ozone is more variable than in spring.

Figure 2C presents seasonal histograms of the number of days in a month at each park where ozone levels exceed the AQI threshold for unhealthy or sensitive groups from 1990 to 2014. The figures omit the density of park-month observations with no AQI warning days for clarity. Few days trigger AQI warnings in winter. The number of park days with AQI warnings increases in spring and fall, and on average, a few parks in each season have warnings every day of the month. Summer months see a large shift in the frequency of parks with warnings on every day of the month. Park observations with AQI warnings on every day of the month in the summertime are as frequent as park observations with AQI warning days for just 1 week.

Columns 3 to 6 of Table 2 present estimates of the effect of visibility on visitation. The FE estimates (columns 3 and 5) suggest that improved visibility has a negligible association with visitation. The FE estimates are subject to similar criticisms of time-varying omitted variable bias, measurement error, and reverse causality as our pollution and visitation estimates. When we instrument for visibility using upwind ozone concentrations (columns 4 and 6), we find a positive association between improved visibility and visitation. However, all visibility coefficients are imprecisely estimated, and the IV regressions have small F statistics, suggesting that the estimate may suffer from weak instrument bias.

We explore the sensitivity of both the visitation response to ozone concentrations and the potential mechanisms for the relationship in the Supplementary Materials. While results are robust to a variety of alternative specifications, additional controls, and IV strategies, we are unable to definitively rule out other unobserved factors that may explain these findings.

DISCUSSION

This study documents that trends in high ozone days at U.S. national parks align with recent regulatory efforts to improve air quality in these areas. However, improvements are more modest than those in major metropolitan areas. While metropolitan areas have seen steady progress since the CAA of 1990, average ozone reductions in national parks have only been apparent since the early 2000s. This improvement in parks aligns with the passage of the EPA Regional Haze Rule in 1999, which sought to improve visual air quality in national parks.

The estimated negative relationship between ozone and park visitation suggests that these air quality improvements benefit the public. Previous research documents society's willingness to pay for environmental improvements at national parks (*30*). Our results support and extend these results, demonstrating that individuals' observed behavior is consistent with their valuing air quality improvements in parks.

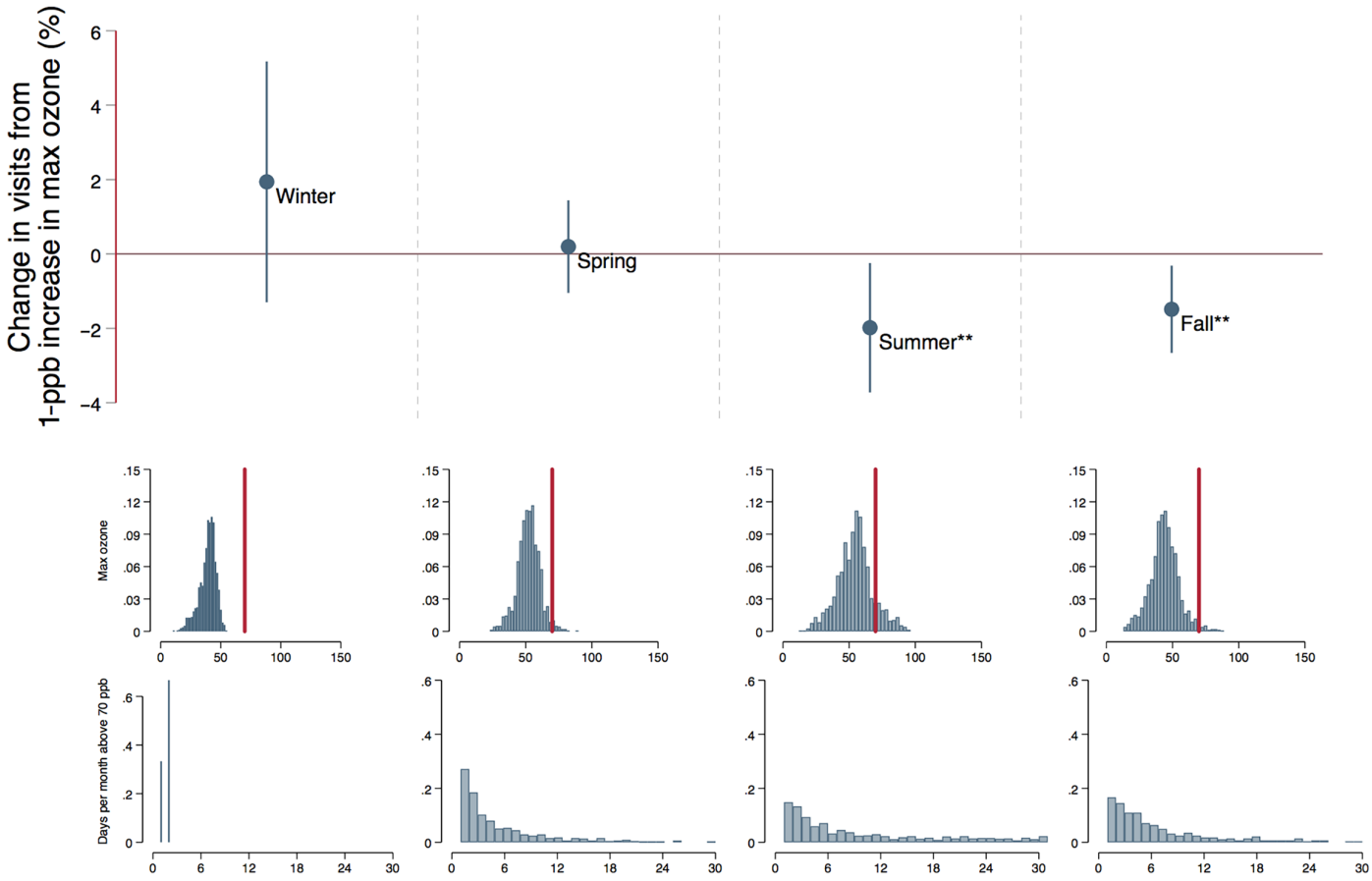

Fig. 2. Seasonal effects of maximum ozone (ppb) on visitation. (**A**) Mean estimate and 95% confidence intervals for the effect of maximum ozone on visitation by season. (**B**) Seasonal histograms of maximum ozone. The red line indicates the 70-ppb threshold. (**C**) Seasonal histograms of average monthly exceedance days. ***P < 0.01, **P < 0.05, and *P < 0.1.

Our results also have implications for human health. Despite the negative association between visitation and ozone, we estimate that around 35% of all visitor days at parks in our sample (284 million visitor days since 1990) occurred when ozone exceeded the 55-ppb "Moderate" AQI threshold. Nearly 9% of visitor days (77 million visitor days) occurred at parks when ozone levels exceeded 70 ppb. A large body of evidence finds that ozone exposure increases hospitalization rates (*26*, *27*), respiratory symptoms (*31*), and mortality (*32*, *33*). These adverse effects from exposure are greater during exercise (*7*). The number of national park visits suggests potentially large human health benefits to further air quality improvements.

MATERIALS AND METHODS

Data sources

Daily national park pollution and weather data are from the NPS Gaseous Pollutant Monitoring Program provided by the NPS Air Resource Specialists. Coverage was limited in many parks, particularly for pollutants other than ozone. We first limited our analysis to national parks with consistent ozone pollution monitoring from 1990 to 2014 (table S1). We further limited our sample to parks for which we had consistent weather and visitation data. Last, we did not include parks for which we could not construct a measure of upwind ozone concentrations. The final data set included the 33 bolded parks in table S1. Parks had, on average, 1.5 ozone monitors in a given year. Most monitors are near visitor centers, with some monitors located in more remote areas of parks.

Comparison metropolitan areas were chosen on the basis of population estimates from the 2010 U.S. Census Bureau. Daily pollution data for cities were from the EPA AirData database, and weather data for metropolitan areas were from the PRISM Climate Group at Oregon State University. We included monitor readings from all counties listed in table S2 to account for air quality trends both in and around city centers.

The final data set used in the trend analysis included the maximum daily 8-hour average ozone readings from any monitor in a park or metropolitan area. Data used for the visitation analysis included monthly national park visitation data from the NPS public use statistics, as well as NPS monitor readings for visibility and other pollutants. County adjacency data were from the U.S. Census Bureau, and we used the pollution transportation matrices from the Air Pollution Emission Experiments and Policy (APEEP) integrated assessment model to determine upwind, intercounty rankings for our atmospheric pollution transport data (*34*).

Pollution trends

Ground-level ozone is a secondary pollutant formed through complex chemical reactions between nitrogen oxides (NO_x), VOCs, and local weather. We control flexibly for the influence of seasonal and local weather conditions in our primary analysis to isolate trends in ozone

Table 2. Estimated impact of exceedance day fraction, maximum visibility, and mean visibility on log visitation from 1990 to 2014. Exceedance day fraction is the fraction of days in a month where maximum daily 8-hour ozone concentrations exceed 70 ppb. Observations decrease when we include visibility readings due to a more limited visibility monitoring by NPS. All specifications include weather controls, park-by-year FEs, and month-of-year FEs. Instrumental variation specifications instrument for each variable using ozone concentrations in upwind counties. Values in parentheses are robust SEs clustered at the park using a bootstrap procedure with 500 replications. ***$P < 0.01$, **$P < 0.05$, and *$P < 0.1$. The Stock-Yogo weak identification critical value for the Kleibergen-Paap F test is 16.38 for a 10% maximal IV bias relative to ordinary least squares.

Exceedance day fraction	−0.845**	−2.693**				
	(0.335)	(1.276)				
Maximum visibility (km)			−0.000236	0.0191		
			(0.000493)	(0.0935)		
Mean visibility (km)					−0.000754	0.0228
					(0.00123)	(0.238)
Observations (N)	5603	5603	4761	4761	4761	4761
Number of parks	33	33	29	29	29	29
Mean days above 70 ppb	1.89	1.89	—	—	—	—
Maximum visibility (km)	—	—	214.50	214.50	—	—
Mean visibility (km)	—	—	—	—	147.20	147.20
Weather controls	Yes	Yes	Yes	Yes	Yes	Yes
Month FE	Yes	Yes	Yes	Yes	Yes	Yes
Park-by-Year FE	Yes	Yes	Yes	Yes	Yes	Yes
IV	No	Yes	No	Yes	No	Yes
Kleibergen-Paap F test	—	33.33	—	5.39	—	4.08
R^2	0.904	0.891	0.910	0.744	0.911	0.809

concentrations due to anthropogenic and other non–weather-related factors.

We estimate the following regression

$$Y_{it} = \sum_{\tau=1991}^{2014} \alpha_\tau 1[t_t = \tau] + \sum_{\tau=1991}^{2014} \gamma_\tau 1[t_t = \tau]\Theta_i + \boldsymbol{W}'_{it}\beta_{\boldsymbol{w}} + \boldsymbol{S}'_{it}\beta_S + \delta_i + \epsilon_{ijt} \quad (1)$$

where Y_{it} is either the maximum 8-hour ozone reading from any monitor in park/city i on date t or an indicator for whether any monitor in park/city i on date t exceeded 70 ppb. An observation reflects the maximum of any reading in the city or park on a given date. $\mathbf{1}[\boldsymbol{t_t} = \boldsymbol{\tau}]$ is an indicator variable for whether date t is in year τ; Θ_i is an indicator variable for whether location i is a national park; $\boldsymbol{W}'_{it}$ are flexible weather covariates that control for the impact of weather on ozone formation, including third-order polynomials of daily precipitation, mean temperature, minimum temperature, and maximum temperature; $\boldsymbol{S}'_{it}$ are flexible seasonality control variables (third-order polynomials of the day of year); and δ_i are park/city fixed effects.

The coefficients of interest are α_τ and γ_t, the annual average ozone or exceedance days relative to 1990 levels in cities and parks, respectively. We identified the average ozone or exceedance day level in cities in year τ as $\bar{Y}_{\text{city},1990} + \alpha_\tau$, where $\bar{Y}_{\text{city},1990}$ is the average ozone level or the number of exceedance days in cities in 1990. Trends for parks are equal to $\bar{Y}_{\text{park},1990} + \alpha_\tau + \gamma_\tau$, where $\bar{Y}_{\text{park},1990}$ is the average ozone level or the number of exceedance days in parks in 1990. We multiplied the estimated trends in ozone exceedance days by 365, so that trends represent the average number of days of ozone exceedance in cities and parks in year τ, respectively. Bootstrapped standard errors (SEs) are robust to heteroscedasticity and clustered at the park/metropolitan area to correct for autocorrelation in the error term and are computed using a bootstrap procedure with 500 repetitions.

Ozone and park visitation

We used two empirical strategies to estimate the impact of ozone on national park visitation. The first model is given by

$$Y_{pym} = \beta X_{pym} + \boldsymbol{\alpha_w} g(\boldsymbol{W}_{pym}) + \boldsymbol{\theta_{py}} + \boldsymbol{\gamma_m} + \epsilon_{pym} \quad (2)$$

where p indexes park, y indexes year, and m indexes month of year. Y_{pym} is the natural logarithm of recreational visits. X_{pym} is our variable of interest, either the monthly average 8-hour daily maximum ozone concentrations, $O3_{pym}$ (maximum ozone), or the fraction of days in a month where the daily maximum 8-hour ozone reading exceeded 70 ppb, Above70$_{pym}$ (exceedance day fraction). $g(\boldsymbol{W}_{pym})$ is a vector of weather controls including a linear spline of temperature with three knots, and average rainfall, humidity, and wind speed. Temperature splines account for nonlinear impacts of weather on visitation and ozone formation (section S1.3). Estimates using alternative functional forms for weather controls can be found in the Supplementary Materials. θ_{py} is the park-by-year fixed effects that control for unobservable covariates in each park year, and γ_m is the month-of-year fixed effects to control for unobserved seasonal effects common across parks. Bootstrapped SEs were robust to heteroscedasticity and clustered at the park/metropolitan area to correct for autocorrelation in the error term and were computed using a bootstrap procedure with 500 repetitions.

Our second empirical strategy is an IV model. We instrument for X_{pym} in Eq. 2 using a first-stage regression given by

$$X_{pym} = \beta O3^{up}_{pym} + \alpha_w g(\boldsymbol{W}_{pym}) + \boldsymbol{\theta_{py}} + \boldsymbol{\gamma_m} + \epsilon_{pym} \quad (3)$$

where $O3^{up}_{pym}$ is our instrument, and all other terms are the same as in (2). The instrument was constructed using monthly average 8-hour daily maximum ozone concentrations from counties upwind of park p. We used the same instrument for both variables to maintain the same variation and ensured that the estimated local average treatment effects are as comparable as possible. We determined upwind counties

using the atmospheric dispersion model embedded in the APEEP integrated assessment model (*34*). The APEEP model includes source-receptor transport matrices that designate the portion any pollutant emitted from one county that is transported to all other counties in the contiguous United States. We used the transport matrix for each county containing a national park to rank all other counties in descending order based on the fraction of one unit of emissions in the given county that leads to an increase in pollution concentrations in the national park county. We then calculated the mean monthly averages of the 8-hour daily maximum ozone concentrations across upwind counties to create $O3^{up}_{pym}$. The second-stage regression in our IV specification is the same as Eq. 2, but where X_{pym} is replaced with its first-stage predicted value $\hat{X}_{pym}$.

We estimated the season-specific impacts of ozone on visitation from Fig. 2 using the following regression

$$Y_{pym} = \gamma_0 + \sum_{S=\mathrm{w,sp,su,f}} \beta_m O3_{pym} \times 1(m \in S) + \boldsymbol{\alpha_w} g(\boldsymbol{W}_{pym}) + \boldsymbol{\theta_{py}} + \boldsymbol{\gamma_m} + \epsilon_{pym} \quad (4)$$

where $1(m \in s)$ is an indicator for whether month-of-year m is in season S. Winter (w) includes December, January, and February; spring (sp) includes March, April, and May; summer (su) includes June, July, and August; and fall (f) includes September, October, and November.

SUPPLEMENTARY MATERIALS

Supplementary material for this article is available at http://advances.sciencemag.org/cgi/content/full/4/7/eaat1613/DC1

Supplementary Text

Fig. S1. Unconditional trends in maximum daily 8-hour pollution and days with maximum daily 8-hour pollution exceeding 70 ppb in large metro areas and national parks.

Fig. S2. Nonlinear impacts of ozone and weather on visitation.

Fig. S3. Seasonal trends in control and outcome variables.

Fig. S4. Estimated impact of exceedance days by AQI category on log visitation from 1990 to 2014.

Table S1. Pollution monitors, visitation, upwind ozone, and weather data by national park and year.

Table S2. Top 20 U.S. cities.

Table S3. Estimated impact of monthly average maximum ozone concentrations (ppb) in national parks on log visitation from 1990 to 2014 (additional specifications).

Table S4. Estimated impact of monthly average maximum ozone concentrations (ppb) in national parks on log visitation from 1990 to 2014 (robustness checks).

Table S5. Estimated impact of alternative monthly average pollution levels in national parks on log visitation from 1990 to 2014.

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