

Urban Heat and Air Pollution

An Emerging Role for Planners in the Climate Change Debate

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This article presents empirical evidence linking recent fluctuations in regional temperatures to enhanced ozone formation within the country's 50 largest metropolitan regions. The results of an analysis of regional climate and ozone formation during the 1990s indicate that annual violations of the national ozone standard were more strongly associated with regional temperatures than with the emissions of regulated ozone precursors from mobile and stationary sources. Based on the results of this analysis, I argue that the air quality management strategies outlined in the Clean Air Act may be insufficient to control ozone formation due to ongoing and unanticipated changes in global and regional climate. I further argue that the emergence of urban heat as a significant air "pollutant" demands a strategic response from the field of urban planning. The article concludes with a discussion of the linkages between urban form and regional temperature and outlines a set of design strategies that have proven successful in mitigating urban heat production.

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The return of record-breaking heat waves to global cities in 2003 once again sparked national interest in the phenomenon of global warming. Evidence in the recent global temperature record of a sustained warming trend is impressive. The 10 warmest years on record have all occurred since 1990. Surpassed only by 1998, 2002 ranked as the second warmest year since 1880, the earliest date for which reliable global meteorological data is available, and by the close of the year, 2003 had displaced 2001 as the third warmest year on record. To date, 19 of the 20 warmest years have occurred since 1980, with the years 1997 through 2003 now accounting for 7 of the 10 of the most extreme annual global temperature anomalies ever recorded.¹

While a small contingent of the global scientific community continues to challenge the theory of a human-induced greenhouse effect, there is little debate over the empirical temperature record of the last several decades. The planet is experiencing a warming trend unprecedented in both its magnitude and rate of increase (Intergovernmental Panel on Climate Change [IPCC], 2001). As even natural periods of sustained warming revealed in the geological record have persisted for hundreds of years, it is reasonable to conclude that the global climate will be characterized by elevated temperatures well into the future. In light of this prospect, there is a critical need for the field of planning to consider the implications of climate change, especially for urban populations.

One issue of paramount concern for North American cities is the role of climate change in regional ozone formation. While media attention has generally focused on the more exotic implications of a warming climate, such as violent weather and the pole-oriented migration of infectious diseases, the most significant impacts are likely to result from an exacerbation of well established urban environmental problems, such as air pollution and water shortages (IPCC, 2001). While the relationship between a warming climate and extended periods of drought is widely acknowledged, less well understood, particularly in the field of planning, are the linkages between temperature and air quality. For many North American cities, however, persistent air quality issues may prove to be the most tangible—and threatening—symptom of a warming world.

This article presents empirical evidence linking recent fluctuations in regional temperatures to enhanced ozone formation within the country's 50 largest metropolitan regions. The intent of this study is to measure the association

between temperature change and urban ozone formation during the 1990s and to contrast this relationship with changes in the ambient concentrations of regulated ozone precursors. Based on the results of this analysis, I will argue that the urban air quality management strategies outlined in the Clean Air Act may be insufficient to control ozone formation due to ongoing and unanticipated changes in global and regional climate. I will argue further that the emergence of urban heat as a significant air "pollutant" demands a strategic response from the field of urban planning. The article concludes with a discussion of the linkages between urban form and regional temperature and outlines a set of design strategies that have proven successful in mitigating urban heat production.

The Persistent Problem of Ozone

The dramatic growth in American automobile use following World War II effected a significant change in both the nature of and regulatory response to urban air pollution. Prior to this time, urban air pollution was largely characterized by the heavy particulate of industrial coal smoke and was generally confined to the downtown cores of America's largest industrial cities. By the late 1940s, however, an unfamiliar haze had descended upon large, auto-oriented regions such as Los Angeles, reducing visibility and inflaming the respiratory tracts of area residents (Elsom, 1992). First associated with the deadly pollution fogs of London, the pervasive irritant was misdiagnosed as a combination of industrial smoke and fog, or "smog," a misnomer that persists to this day. Science would soon reveal that the pollutant was not an industrial particulate but a gaseous oxidant known as ozone. The onset of this pollutant would prove to be one of many unforeseen consequences of America's burgeoning love affair with the automobile.

The emergence of mobile source air pollution altered the nature of urban air quality management in a number of important ways. First, as a mobile combustion plant, the automobile creates a transboundary pollution issue any time it crosses state lines. Under the broad parameters of the federal Commerce Clause of the U.S. Constitution (art. I, § 8, cl. 3), such interstate movement of mobile source pollution enables the federal government to play a role in regional air quality management. More important in this regard, secondary pollutants such as ozone—those that are not emitted directly from tailpipes but develop within the atmosphere—are capable of being transported great distances (Elsom, 1992). As a result, the diffuse nature of ozone often creates a transboundary problem even in the

absence of interstate vehicle travel. Thus, the regionalized nature of both the source and the effect of tropospheric ozone created the need for a multistate approach to air quality management.

The development of a national air quality control program has greatly diminished the significance of land use planning in urban air quality management. Prior to the enactment of the original Clean Air Act (1963), municipal governments generally employed exclusive-use zoning to separate noxious industrial activities from residential and commercial zones; Los Angeles, for example, created one of the first exclusive industrial activity zones in 1909 (Scott, 1995, p. 76). Empowered with no such land use authority, the federal government has relied heavily upon a technological "end of the pipe" approach to urban air quality management. In so doing, rather than emphasize the demand-side approaches of transit-oriented development or congestion pricing, the federally administered air pollution control program has required automobile manufacturers to develop increasingly effective vehicle emissions control technologies (Schlesinger & Horowitz, 1998).

This technological emissions control program, first initiated with the Motor Vehicle Air Pollution Control Act of 1965, has been greatly successful in reducing pollutant emissions per mile of vehicle travel. For example, average per-mile emissions of volatile organic compounds (VOC), one of the two principal ozone precursors emitted by automobiles, have been reduced by approximately 90% since the 1968 model year (U.S. Environmental Protection Agency [EPA], 2000b). Yet, due in part to the dramatic and ongoing increase in annual per capita vehicle miles of travel during this period, as well as population growth, the total reduction in mobile-source emissions of ozone precursors has not been as substantial as anticipated, a trend further exacerbated by the recent decline in fleet average fuel efficiency associated with sport utility vehicles. It is one of the great ironies of recent environmental history that despite tremendous technological progress in reducing ozone precursor emissions from both mobile and stationary sources, almost 120 million Americans reside today in regions classified as nonattainment areas for the ozone standard (EPA, 2002).

An even greater irony, perhaps, is that despite a reduction in point and mobile source emissions of ozone precursors, the number of high ozone days in large metropolitan regions has not consistently declined.² Figure 1 illustrates the average number of days per year in which the national Air Quality Index (AQI) for ozone exceeded 100—a level determined to be unhealthy—in the 50 largest (by population) cities of the U.S. during the decade of the 1990s.³ The figure indicates that urban ozone formation has fluctuated

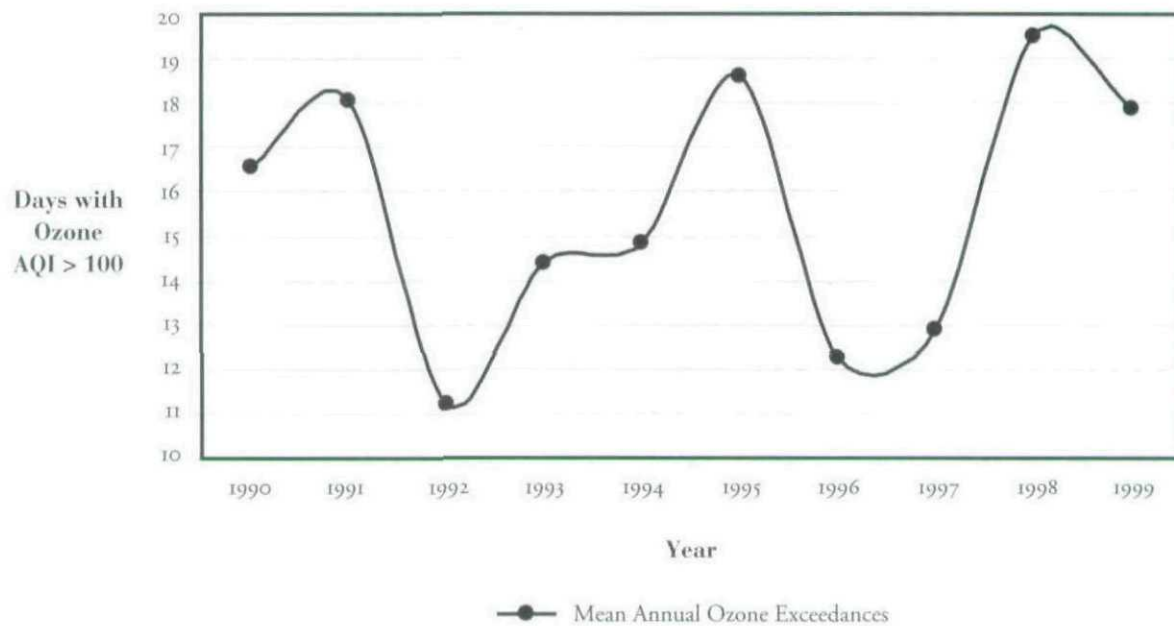


Figure 1. Mean number of annual ozone exceedances in 50 largest U.S. metropolitan regions, 1990–1999.
Source: EPA (2000a)

greatly from year to year, and that the average number of unhealthy air quality days in the final year of this decade was approximately 10% greater than in 1990.⁴

This is a discouraging outcome for two reasons. First, the primary goal of the federal Clean Air Act is to improve air quality on an annual basis. Ten years after the passage of the Clean Air Act Amendments of 1990, this goal was not achieved for ozone air pollution within large metropolitan regions of the U.S. Second, as noted above, total emissions of ozone precursors from both stationary and mobile sources have not fluctuated greatly from year to year, but rather have declined or remained stable since 1990. The EPA reports that for all metropolitan statistical areas, emissions of VOC have decreased by 10% since 1990, while emissions of nitrogen oxides have remained relatively stable (EPA, 2000a). Thus, the ozone problem has persisted despite attempts to address its sources.

The trends identified above serve to illuminate the central question of this study: Why has urban ozone formation failed to show a significant reduction in recent years despite a reduction in the emission of ozone precursors? As the remainder of this article argues, the answer to this question has profound implications for the nature of urban air quality management in this country and others of the developed world. To help illustrate why our successful emissions control program has failed to produce clean air

within most large metropolitan regions, a brief overview of the process of ozone formation and air quality management is needed.

Fundamentals of Ozone Formation and Control

Ozone is a colorless and odorless gas that is formed in the presence of sunlight through the chemical interaction of nitrogen oxides (NO_x) and VOC, two classes of pollutants produced from the combustion of fossil fuels. Ozone consists of three atoms of oxygen (O₃) and, like its close cousin diatomic oxygen (O₂), is present in several layers of the atmosphere. When found in the stratosphere, the layer extending from between 10 and 50 kilometers above the earth's surface, ozone acts to absorb harmful ultraviolet radiation emitted from the sun. It is this stratospheric ozone layer that has been depleted in recent decades by the release of chlorofluorocarbons and other compounds, and which is essential to life on earth. By contrast, tropospheric ozone, the same molecule when found in the lowest layer of the atmosphere, is regarded as an air pollutant due to its deleterious effects on human health, vegetation, and buildings (Nebel & Wright, 1998).

First measured in Los Angeles during the 1940s, high ambient concentrations of urban ozone were initially associated with irritation of the throat and eyes. The more recent epidemiological literature has linked ozone to a range of serious health effects, including an increased prevalence of acute asthma, reduced cardiopulmonary function, aggravated respiratory disease, and premature mortality among both children and adults. A number of studies in North America and Europe have documented a sharp increase in hospital admissions on high smog days, particularly among children, and studies focusing on the potential long-term health effects of urban air pollutants, including ozone, have documented a potential reduction in life expectancy on the order of years (Touloumi et al., 1997). In addition to human health concerns, ozone air pollution accelerates the deterioration of buildings and is responsible for annual crop losses in this country valued in the hundreds of millions of dollars (EPA, 2000b).

The photochemistry of the ozone formation process has important implications for urban air quality management. Fueled by the availability of both VOC and NO_x, ozone formation can be slowed through a reduction in either precursor. As a result, technological emissions control devices designed to reduce one or both of these precursors have been developed and installed in automobiles and a range of industrial processes. The current generation of catalytic converters, for example, is engineered to reduce both NO_x and VOC from vehicle exhaust, providing a straightforward approach for managing ozone formation. Yet, as noted above, this approach has been frustrated by growth in both per capita vehicle travel and energy consumption, ongoing trends that have greatly increased the quantity of emissions to be controlled.

A second important limitation of technological emissions controls is that such devices are incapable of reducing natural or "biogenic" sources of ozone precursors. In addition to VOC emitted by automobiles and industrial sources, various species of trees produce natural forms of VOC, such as isoprene (Sillman, 1999). The effects of temperature change on biogenic precursor emissions can be substantial. For example, in modeling the effects of temperature change on ozone formation in the Atlanta, Georgia, region, Cardelino and Chamedies (1990) found a 2°C rise in temperature to be associated with an approximately 45% increase in biogenic isoprene emissions (p. 13,977). Similarly, the thermal decomposition of an atmospheric compound known as peroxyacetyl nitrate, or PAN, can also promote ozone formation. With rising ambient temperatures, the PAN compound decomposes into peroxyalkyl radicals and nitrogen dioxide. Through the liberation of sequestered nitrogen dioxide, the chemical

decomposition of PAN increases the atmospheric reservoir of NO_x (Cardelino & Chamedies, 1990). The generalized role of biogenic precursors, anthropogenic precursors, and ambient heat in the ozone formation process is illustrated in Figure 2.

The sensitivity of ozone formation to ambient temperature suggests an important and largely unexamined hypothesis concerning recent trends in urban air quality: Divergent trends in ozone exceedances and precursor emissions within large metropolitan regions are primarily attributable to temperature fluctuations during the 1990s.⁵ If correct, this relationship holds important implications for the viability of our current national approach to urban air quality management. Most significantly, the identification of ambient heat as a principal driver of urban air quality would suggest the need for policies designed to offset the rise in urban temperatures produced through land use practices and energy consumption. To test this hypothesis, I investigated the association between annual temperature fluctuations and ozone formation in the country's largest metropolitan regions and considered the implications of this relationship for future air quality management and the field of urban planning.

Recent Trends in Urban Climate and Ozone Formation

The ozone mitigation provisions outlined in the Clean Air Act Amendments of 1990 constitute the most comprehensive environmental legislation ever passed by the U.S. Congress (Bryner, 1993). More than a decade after the passage of this landmark legislation, there is a need to assess the success of our national air quality management program in reducing tropospheric ozone. Such an assessment should address three important questions: (1) During the 1990s, how effective were VOC and NO_x control strategies in reducing total annual emissions of these pollutants in the country's largest metropolitan regions? (2) During this same period, how effective were these strategies in abating the average number of high ozone days per year? (3) How closely did annual violations of the ozone standard correlate with fluctuations in regional temperatures?

The discussion that follows attempts to answer these questions using air quality and climatological data obtained from the EPA and the National Oceanic and Atmospheric Administration (NOAA). The results of this study are intended to illustrate the relative significance of a single climatological variable—temperature—to ozone control strategies within major metropolitan regions of the U.S.

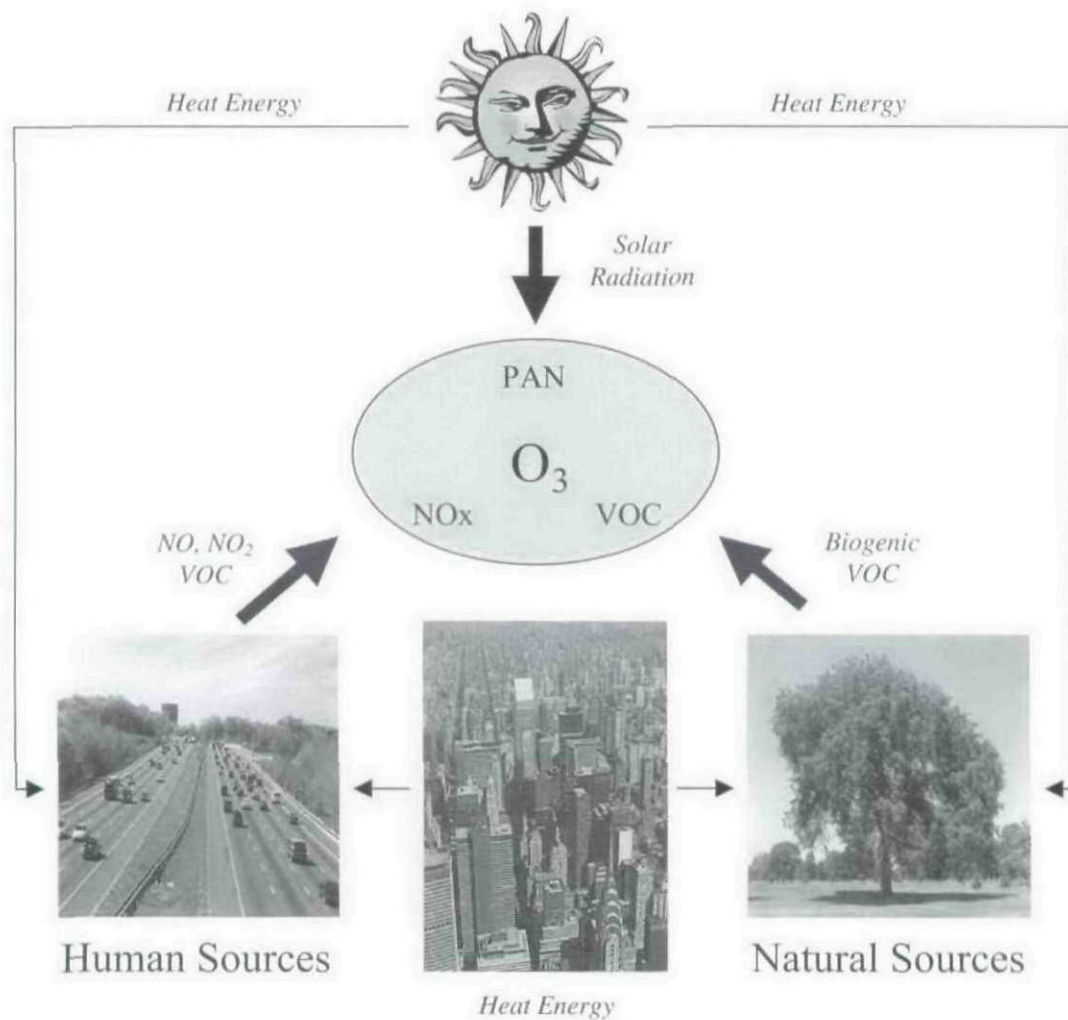


Figure 2. The ozone formation process.

While climate is generally regarded as an environmental variable beyond the control of public policy, there is a mounting body of evidence to suggest that metropolitan land use activities and energy consumption produce a measurable effect on both regional and global climates (e.g., IPCC, 2001; Saitoh et al., 1996). If true, climate-responsive design strategies may provide an additional policy tool to mitigate regional ozone formation. This analysis gauges the potential efficacy of a climate-based ozone control strategy relative to our current VOC and NO_x abatement programs.

The VOC and NO_x emissions data used for this analysis were obtained from the EPA's National Emissions Inventory (NEI). The NEI combines emissions data recorded at all major industrial sources with mobile source emissions estimates modeled by state and regional trans-

portation planning agencies. Annual emissions of VOC and NO_x are two of the six "criteria," or nationally regulated, pollutants tracked by the NEI. For this analysis, emissions data were obtained at the county level and aggregated by the U.S. Census Bureau's metropolitan statistical area designations into regional estimates. As noted above, the metropolitan regions selected were ranked according to population estimates derived from the 2000 U.S. census. A list of surveyed regions is provided in Figure 3.

Figure 4 presents the annual trends in emissions of VOC within the 50 metropolitan regions surveyed. Depicted on the primary (left) y-axis, the curve marked with triangles illustrates the regional mean annual tonnage of VOC emissions. In order to facilitate visual comparisons between the two trends, the mean annual ozone exceedance data presented in Figure 1 is plotted along the second-

Metropolitan Region (2000 population in millions)

1. New York, NY (21.2)	18. St. Louis, MO (2.6)	35. New Orleans, LA (1.3)
2. Los Angeles, CA (16.4)	19. Baltimore, MD (2.6)	36. Salt Lake City, UT (1.3)
3. Chicago, IL (9.1)	20. Denver, CO (2.6)	37. Greensboro/Winston Salem, NC (1.3)
4. San Francisco/Oakland, CA (7.0)	21. Tampa/St. Petersburg, FL (2.4)	38. Austin, TX (1.2)
5. Philadelphia, PA (6.2)	22. Pittsburgh, PA (2.4)	39. Nashville, TN (1.2)
6. Boston, MA (5.8)	23. Portland, OR (2.3)	40. Providence, RI (1.2)
7. Detroit, MI (5.5)	24. Cincinnati, OH (2.0)	41. Raleigh/Durham, NC (1.2)
8. Dallas/Ft. Worth, TX (5.2)	25. Sacramento, CA (1.8)	42. Buffalo, NY (1.2)
9. Washington, DC (4.9)	26. Kansas City, MO (1.8)	43. Memphis, TN (1.1)
10. Houston, TX (4.7)	27. Milwaukee, WI (1.7)	44. West Palm Beach, FL (1.1)
11. Atlanta, GA (4.1)	28. Orlando, FL (1.6)	45. Jacksonville, FL (1.1)
12. Miami, FL (3.9)	29. Indianapolis, IN (1.6)	46. Rochester, NY (1.1)
13. Seattle, WA (3.5)	30. San Antonio, TX (1.6)	47. Grand Rapids, MI (1.1)
14. Phoenix, AZ (3.3)	31. Norfolk/Virginia Beach, VA (1.6)	48. Oklahoma City, OK (1.1)
15. Minneapolis/St. Paul, MN (3.0)	32. Las Vegas, NV (1.6)	49. Louisville, KY (1.0)
16. Cleveland, OH (2.9)	33. Columbus, OH (1.5)	50. Richmond/Petersburg, VA (1.0)
17. San Diego, CA (2.8)	34. Charlotte, NC (1.5)	

Figure 3. Fifty most populous U.S. metropolitan regions.

Note: Hartford, CT, was omitted from the analysis due to the unavailability of complete temperature data.

ary (right) y-axis. The results of this simple trend analysis indicate that significant progress was made during the 1990s in reducing mean annual emissions of VOC. Following a period of modest increase in the first several years of the decade, VOC emissions exhibited a consistent and substantial reduction from the years 1994 through 1999. These results clearly validate the success of our national air quality management program in reducing stationary and mobile-source emissions of this critical ozone precursor.

Given this clear progress in VOC control, it is to be expected that ozone exceedances would exhibit a declining trend as well. However, the ozone exceedance data presented in Figure 4 appear to tell a different story. Rather than trending downward or leveling off, the average number of ozone violations exhibited a sharp increase in the latter years of the 1990s. With the exception of a few years, the ozone observations do not appear to correspond closely to metropolitan VOC emissions during the 10 years examined. As noted in the figure, the statistical correlation between VOC emissions and ozone exceedances was not found to be significant. These findings provide statistical evidence that our national VOC control program has not proven successful in mitigating ozone exceedances in large cities during the most recent decade for which complete data is available.

Annual emissions of nitrogen oxides were also found to correspond weakly with trends in metropolitan ozone formation. Figure 5 plots trends in mean NO_x emissions alongside mean ozone exceedances for the same metropolitan areas surveyed in Figure 4. In contrast to trends in VOC emissions, emissions of NO_x revealed no sustained downward patterns during the 1990s. Consistent with national trends, annual metropolitan emissions of NO_x appear to have leveled off by the middle of the decade, and hence, correspond poorly to the more significant fluctuations during this period in ozone exceedances. Similar to the preceding analysis, and as indicated by the correlation coefficient of -0.25 , these results do not provide evidence of a statistically significant relationship between anthropogenic emissions of ozone precursors and metropolitan ozone formation.

The final question to be addressed by this analysis concerns the relationship between regional temperatures and aggregate trends in metropolitan ozone formation. To address this issue, mean temperature measurements for the typical "ozone season" (May through September) were obtained from NOAA's National Climatic Data Center. As part of the National Environmental Satellite, Data, and Information Service, NOAA maintains an extensive database of regional and global temperatures obtained through

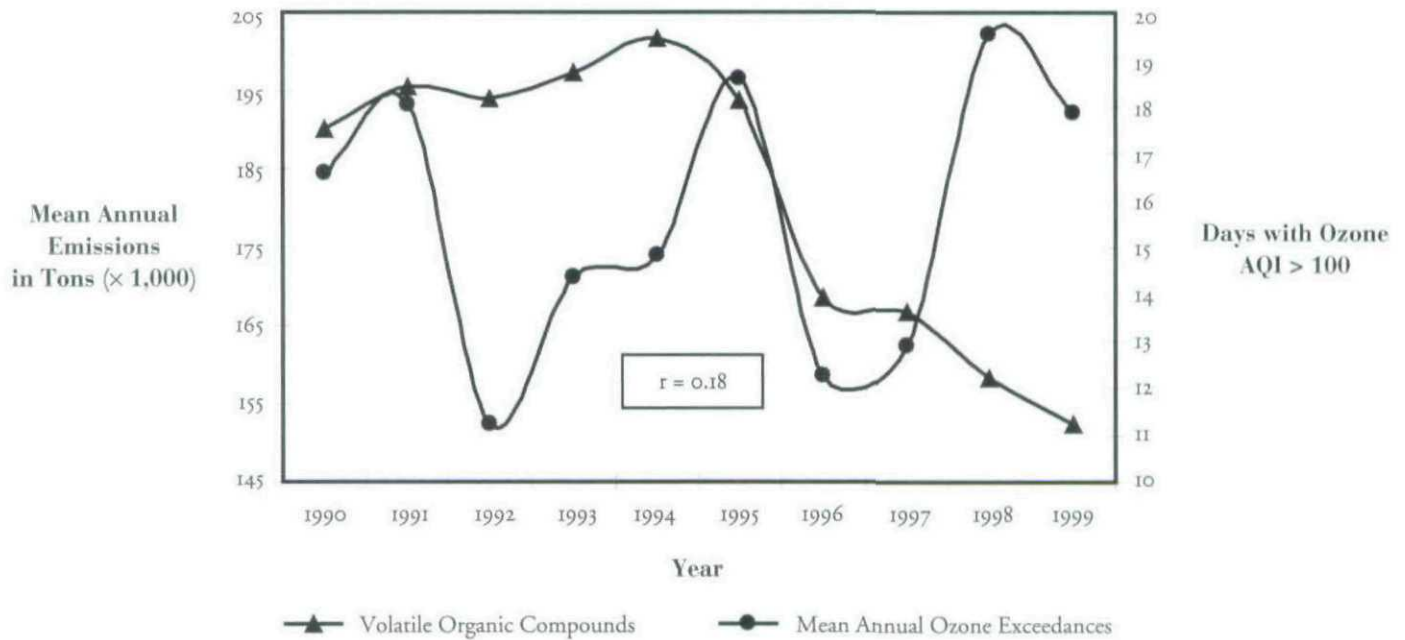


Figure 4. Annual trends in metropolitan VOC emissions and ozone exceedances, 1990–1999.

Source: EPA (n.d.b)

Note: The reported correlation coefficient was not found to be significant at the $p < .05$ level.

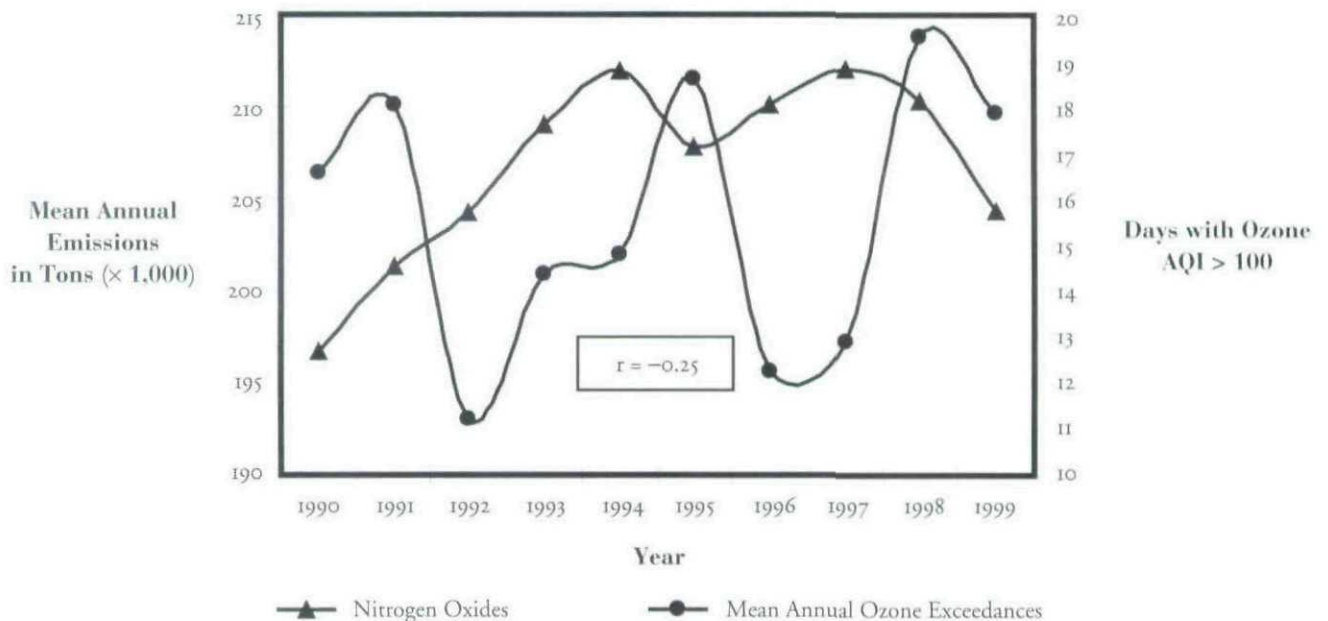


Figure 5. Annual trends in metropolitan NOx emissions and ozone exceedances, 1990–1999.

Source: EPA (n.d.b)

Note: The reported correlation coefficient was not found to be significant at the $p < .05$ level.

a network of climatological stations and satellites. Mean seasonal temperatures recorded in degrees Celsius for the years 1990 through 1999 were obtained for each of the metropolitan areas included in the analysis. The results of this metropolitan temperature trend analysis are presented in Figure 6.

The metropolitan temperature observations manifest a surprisingly strong correlation with mean annual ozone exceedances ($r=0.86$; $p < 0.001$). Mirroring both national and global climatological trends, the years of 1991, 1995, and 1998 were found to be among the warmest of the decade in large U.S. cities, and, in the context of global trends, 3 of the 10 warmest years on record. In close correspondence with these peaks in temperature, the mean number of ozone violations in these regions reached peak levels during each of these years. Likewise, the relatively cool years of 1992, 1996, and 1997 are shown to correspond with the lowest annual ozone exceedance observations. Overall, the graphical covariation between annual temperature and ozone formation appears to be significantly stronger than that observed between ozone exceedances and either VOC or NO_x emissions.

Based on the results of this analysis, annual emissions of both VOC and NO_x within the sampled regions were not found to have a statistically significant correlation with annual ozone exceedances. This outcome raises important

questions concerning the viability of our current approach to urban air quality management. Despite evidence that anthropogenic ozone precursors have either stabilized or declined in the first decade following the enactment of the Clean Air Act Amendments of 1990, these improvements have not been associated with significant reductions in annual urban ozone exceedances. In light of the billions of dollars spent annually on Clean Air Act compliance, there is a critical need to assess the long-term potential for VOC and NO_x mitigation strategies alone to adequately control ozone formation and the many associated health impacts.⁶ Would the diversion of a proportion of these resources to a climatologically-oriented control strategy yield more successful results? The concluding section of this article seeks to address this question in some detail.

Cooling Cities Through Physical Planning

The preceding analysis confirms the existence within the recent meteorological record of a well established relationship between ambient temperature and ozone formation. In light of the sensitivity of urban air quality to ambient heat, urban climate modification may provide a viable approach to ozone abatement. Even so, can regional

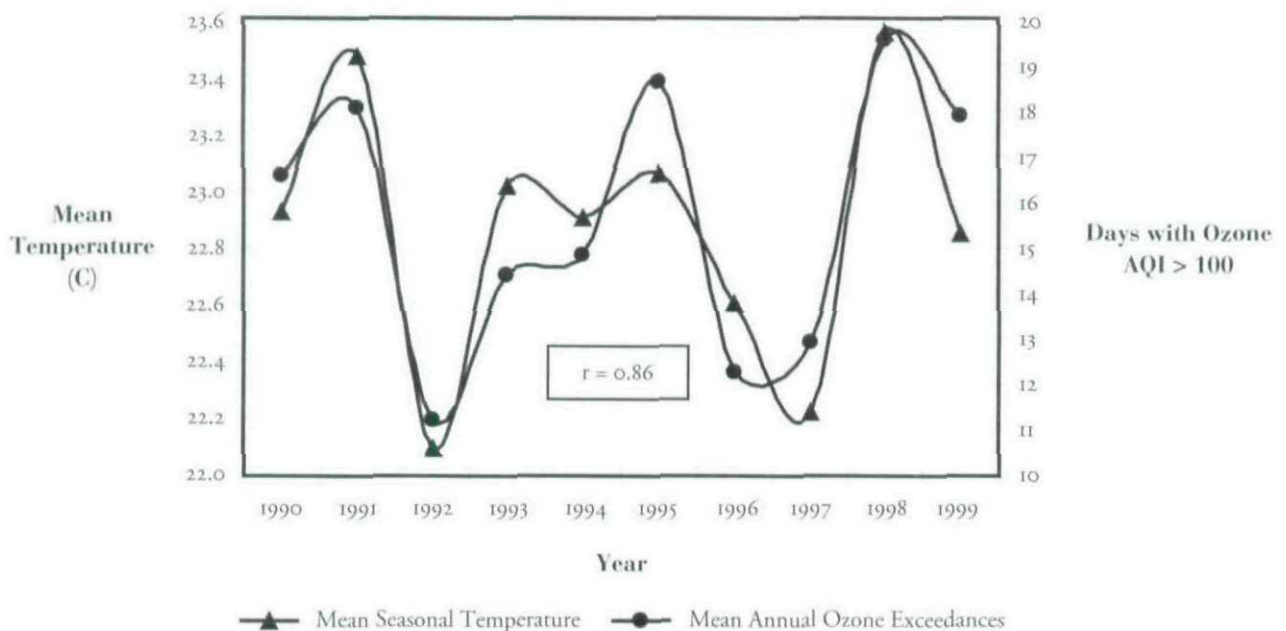


Figure 6. Annual trends in average warm season temperature and ozone exceedances, 1990–1999.

Source: NOAA (n.d.)

Note: The warm season temperatures reported are for the months of May through September. The reported correlation coefficient was found to be significant at the $p < .05$ level.

climates be altered through physical planning? The answer to this question depends upon the scale of the phenomenon to be addressed. At the global level, most climatologists believe the process of global warming ultimately can be reversed through a substantial reduction in the emissions of greenhouse gases. Yet, due to the prolonged atmospheric residence times of many greenhouse gases—as long as 200 years for carbon dioxide—the planet will likely continue to warm for many decades despite corrective measures (IPCC, 2001). A second critical obstacle to global warming mitigation is the difficulty of achieving effective coordination among the global community—a challenge that has proven insurmountable since the drafting of the Kyoto Protocol.

Despite these challenges at the global level, there is a compelling body of evidence to suggest that the physical design of cities has a direct and significant influence on regional warming phenomena. Through a climatological mechanism known as the “urban heat island effect,” urbanized regions have been shown to exhibit average summer temperatures 3.5 to 4.5°C higher than adjacent rural areas (U.S. Department of Energy [DOE], 1996), a localized warming trend that is increasing in many large cities by as much as 1.1°C per decade (McPherson, 1994, p. 151). In contrast to the greenhouse mechanism of global warming, the urban heat island effect is driven not by the emission of greenhouse gases but rather by the physical alteration of the natural landscape and the release of waste heat produced through energy consumption. A second critical distinction between global and regional warming phenomena pertains to the rate and magnitude of the effect associated with each process. While global warming is projected to increase air temperatures by approximately 1.5–6°C over the next 100 years (IPCC, 2001), many large metropolitan regions have already experienced this magnitude of warming relative to adjacent rural areas. In light of these facts, the mitigation of regional heat islands may provide an effective strategy for moderating ongoing warming trends and associated air quality problems within large cities.

While still formative, the literature on urban heat islands identifies at least three physical planning strategies that could be employed to cool cities measurably over a period as short as 10 to 20 years. These strategies include (1) the use of highly reflective paving and roofing materials; (2) the preservation and recultivation of regional forest canopy; and (3) the reduction of waste heat from energy production and consumption.

Increasing Surface Reflectivity

Highly reflective paving and roofing materials can offset heat gain through two mechanisms. First, the min-

eral-based construction materials used in cities have a much greater “heat capacity” than the natural ground cover, enabling these materials to absorb, store, and reemit more thermal energy from the sun and atmosphere than vegetation. Second, many urban surfaces have a much lower reflectivity, or albedo, than natural surfaces, reducing the efficiency with which these materials can reflect away incoming solar radiation (Oke, 1987). Given these relationships, the use of more highly reflective building materials is a promising strategy for cooling metropolitan regions. To assess the potential for this approach, researchers at the DOE and the Lawrence Berkeley Laboratory developed a climatological model to simulate the impact of albedo enhancement on temperature. The results of this model showed that the use of highly reflective roofing and paving materials within the Los Angeles basin could reduce maximum summer temperatures by 1.5°C; the model showed cooling benefits of up to 3°C when a reforestation strategy was combined with albedo enhancement (Rosenfeld et al., 1998, p. 53). The great appeal of urban albedo enhancement is that it can be achieved incrementally over time and at little additional cost through the regularly scheduled resurfacing of streets, parking lots, and roofs.

Increasing Tree Canopy Cover

A second approach to heat island mitigation that is widely supported in the literature is tree preservation and reforestation. In addition to shading ground surfaces from incoming solar radiation, trees and other vegetation utilize energy from the sun to convert water to water vapor. Through this process of evapotranspiration, trees dissipate solar energy that would otherwise be absorbed at the surface and reemitted to the atmosphere as sensible heat energy. Trees thus provide a natural cooling mechanism that is lost through the incremental development of metropolitan regions (Oke, 1987).

The potential for extensive deforestation to elevate regional temperatures is well documented. In an analysis of residential development and surface heat production, Stone and Rodgers (2001) found each quarter-acre increment in single-family lot size in Atlanta, Georgia, to be associated with an average increase in surface heat production of over 30%, a relationship that was primarily attributable to a greater area of deforestation per residential parcel within low-density areas. Similar to albedo enhancement, tree preservation and reforestation are strategies that may be adopted to offset heat production within both new and existing development and yield a range of additional social and ecological benefits, such as stormwater management, wildlife habitat, and aesthetic enhancement.

Reducing Waste Heat

A final important driver of urban warming is the emission of waste heat from industrial processes, mobile sources, and buildings. In addition to the particulate and gaseous air pollutants produced through fossil fuel combustion, great quantities of waste heat are released to the atmosphere from smokestacks and tailpipes. Even more significant within high-density regions can be the heat that is mechanically removed by air conditioning systems from houses and buildings. While the magnitude of waste heat emissions varies widely among and within cities, in some instances the flux of artificial energy can be quite substantial. For example, in comparison to an average solar input of 106 watts per square meter (W/m^2) within central London, McGoldrick (1980) found the flux of waste heat energy to range between 100 and 234 W/m^2 . Similarly, Landsberg (1981) found the flux of anthropogenic heat within parts of Manhattan, New York, to register over 600 W/m^2 .

Design-based approaches to waste heat reduction include energy-efficient building design and transit and pedestrian supportive development. In addition to reducing the quantity of sensible heat energy emitted to the atmosphere, the use of reflective construction materials and the strategic placement of trees around buildings can significantly reduce energy consumption for climate control. For example, Meier (1991) found the placement of mature shade trees to the south and west of single-family homes to offset cooling costs by 25 to 80% during the summer months. Reductions in vehicle engine combustion can also offset regional waste heat emissions. As concluded by a number of studies, moderate to high levels of population and employment density, land use mix, and the provision of transit facilities—the principal elements of transit supportive development—can significantly reduce total auto travel and emissions (e.g., Frank et al., 2000; Johnston & Ceerla, 1995).

The Effectiveness of Design-Based Strategies

While the relationship between the physical characteristics of cities and heat island formation is well established, less well understood is the potential for urban cooling to abate ozone formation. Would a reduction in ambient temperatures of a few degrees Celsius yield measurable improvements in regional air quality and, if so, at what cost? As no city has yet developed a comprehensive heat island reduction program, little direct evidence exists to support a design-based approach to air quality management. Nevertheless, the results of computer simulations designed to model the effects of various development scenarios on air temperature and ozone formation suggest

that modest reductions in heat island formation could produce significant reductions in ozone formation. For example, in their analysis of heat island formation and biogenic emissions, Cardelino and Chamedies (1990) found an increase in ambient temperatures of 2°C to be associated with an increase in regional ozone concentrations from 116 to 140 parts per billion—placing the region in violation of the national one hour standard of 120 parts per billion (p. 13,977).

In the most comprehensive assessment of this question, Rosenfeld et al. (1998) developed a model to quantify the influence of an aggressive albedo enhancement and tree planting campaign on climate and ozone formation in the Los Angeles, California, region. As referenced above, this study found that the planting of 11 million trees and the use of high albedo roofing and paving materials throughout the city were associated with a reduction in summertime maximum temperatures of approximately 3°C. Through the development of an ozone formation and transport model, this cooling effect was found to be associated with a 12% reduction in regional ozone formation. Significantly, the air quality benefits of this modeled program were found to surpass those achievable through California's ongoing programs to phase in cleaner fuels and low to zero emission vehicles. By comparison, these programs were found to reduce regional ozone formation over a 20-year period by 5% and 2 to 4%, respectively (Rosenfeld et al., 1998, pp. 53, 59).

In addition to the magnitude of smog reduction benefits achievable through heat island reduction, these physical design strategies are highly cost effective. Rosenfeld et al. (1998) found the reduction in air temperature and ozone formation described above to be associated with approximately \$535 million in annual energy and public health savings for the Los Angeles region (p. 55). Based on the results of similar analyses conducted in a range of climatological regions, the authors estimated the national annual benefits of albedo enhancement and tree planting to be roughly \$5 billion (Rosenfeld et al., 1998, p. 60). Replaced at the time of routine maintenance (typically every 20 years), the additional costs of high albedo roofing and paving materials were estimated to be less than \$30 per 100 m^2 (Rosenfeld et al., 1998, p. 56).⁷ While Rosenfeld et al. did not assess the costs of tree planting and maintenance, many other studies have found tree planting to be cost effective. In a recent study of tree benefits in Davis, California, Maco and McPherson (2003) found the annual benefits of trees to exceed planting and maintenance costs by a ratio of 3.8 to 1, with about 40% of these benefits attributable to energy savings and reduced air pollution (p. 92). If this ratio holds true for the Los Angeles region, a

large-scale tree planting and albedo enhancement program would be expected to generate environmental and property value benefits far in excess of costs over a 10- to 20-year period.

Discussion and Conclusions

The establishment of a strong association in the recent meteorological record between regional air temperature and urban ozone formation suggests an emerging role for physical planning in the process of air quality management. Generally viewed as a long-term strategy for environmental remediation, environmentally-responsive design may prove to be the most viable approach in the near term for abating urban warming trends and associated air quality problems. Due to the long atmospheric residence times of greenhouse gases, a reduction in current emissions through technological controls would be unlikely to yield measurable results for many decades. In contrast, changes to the physical characteristics of cities can produce cooling benefits at the time of implementation. If aggressively pursued, the installation of highly reflective roofing and paving materials, the preservation and replanting of urban forests, and the minimization of waste heat emissions through energy conservation can produce substantial cooling benefits over the period of a decade or two—far shorter than the period required for benefits to accrue under the Kyoto Protocol.

The development of an aggressive warming mitigation program will require a number of institutional changes at the federal, state, and local levels of government. Most important, the scope of the EPA's regulatory mandate should be broadened to encompass high levels of ambient heat. Specifically, ambient heat should be classified by the EPA as an air pollutant and made subject to the range of regulatory provisions established by the Clean Air Act Amendments of 1990.⁸ Similar to NO_x and VOC, anthropogenically generated heat poses a threat to public health through two mechanisms. First, as explored in detail herein, high levels of ambient heat elevate atmospheric concentrations of ozone precursors. As the primary sources of these heat-sensitive precursors are biogenic in nature, traditional end of the pipe approaches to controlling these compounds are proving to be insufficient and are unlikely to yield measurable improvements in the foreseeable future. Second, high levels of ambient heat pose a direct and rising threat to urban populations in the form of heat stress and heat stroke. As reported by the NOAA (2000), ambient heat is now responsible for more annual fatalities in the U.S. than any other form of catastrophic weather.

The classification of heat as a regulated pollutant could advance the goal of improved urban air quality in a number of ways. First, the incorporation of heat mitigation into regional air quality management strategies would equip metropolitan areas with an additional tool for controlling ozone. For regions characterized by high levels of biogenic precursor concentrations, a balancing of traditional emissions controls with heat mitigation strategies may prove more cost effective over the medium to long term than conventional strategies targeting anthropogenic precursor emissions alone. Second, the formal classification of ambient heat as a pollutant would enable metropolitan regions to apply for federal pollution control funding available through the Transportation Equity Act for the 21st Century (TEA-21).⁹ If air quality benefits can be demonstrated, metropolitan regions might be permitted to apply flexible monies available through TEA-21 toward road resurfacing projects employing high albedo paving materials. In addition, street tree planting programs and the use of porous paving materials may also qualify for federal sources of pollution control funding.

A second institutional change that would promote urban heat mitigation is the incorporation of climate-responsive design strategies into state implementation plans (SIPs). The state's air quality control plan for a period of 5 to 7 years, SIPs identify the range of air pollution control programs that will be adopted within urban areas to conform to the National Ambient Air Quality Standards. While SIPs have traditionally relied upon technological emissions controls and transportation demand management programs to combat ground level ozone, the EPA has recently developed a protocol to enable states to receive air quality "credit" for the adoption of land use strategies demonstrated to improve regional air quality. Categorized as "sustainable" or "smart growth" land use by the EPA, transit and pedestrian oriented development projects may be substituted within SIPs for traditional emissions control strategies if the resulting air quality benefits are believed to be equivalent or greater than those resulting from conventional control methods (EPA, 2001). Presumably, land use practices demonstrated to reduce ozone through heat island abatement would also qualify for SIP credits.

A final set of institutional changes needed to support a design-based heat island and ozone mitigation program entails modifications to local land development regulations. As concluded by Stone and Rodgers (2001), the size and configuration of single-family residential parcels are strongly related to the quantity of excess heat emissions per lot. As a result, zoning and subdivision regulations governing the lot size, permissible area of impervious cover,

neighborhood block size, and number and placement of trees provide a set of well established planning tools that may be modified and aggressively enforced to offset the thermal impacts of new and existing development.

In the interest of promoting regional cooling and associated environmental benefits, a number of urban forestry organizations and local governments have initiated ambitious tree planting campaigns. The Trees Atlanta organization, for example, has planted over 20,000 trees in the Atlanta region since 1985 (Trees Atlanta, n.d.). Along similar lines, the City of Los Angeles is replacing nearly two million square meters of paved schoolyards with greenspace over the next 10 years (EPA, n.d.a).

Perhaps the most promising heat abatement programs entail the revision of building codes to require or encourage the use of high albedo roofing and paving materials. The California Energy Commission, for example, has implemented a Cool Roof Retrofit Program, through which developers may receive a rebate of \$0.05–0.15 per square foot for the use of highly reflective roofing materials. The City of Highland, Utah, has developed a “town center” overlay district, in which reflective paving materials and strategic tree planting are required. Also promising are a large number of “green building” projects designed to offset surface warming and energy consumption. One of many ongoing projects, the installation of a 32,000 square foot roof garden atop Chicago’s City Hall is projected to reduce annual energy bills by over \$4,000. While small in scale, the growing number of green buildings provides a much needed set of field experiments to validate the meteorological and air quality benefits of climate-responsive design (EPA, n.d.a)

In closing, it is important to emphasize that physical planning does not constitute a comprehensive solution for managing climate change and climate-induced air pollution. Driven by global patterns of energy consumption and land use, the phenomenon of global warming may be stabilized over the long term only through a widely coordinated reduction in the emission (or increase in the sequestration) of greenhouse gases. Yet changes to the physical structure and surface character of cities provides a near-to medium-term strategy for adapting to global changes in climate through actions implemented at the regional level. Oriented toward both the spatial and technological bases of climate change, climate-responsive design embodies a viable and pragmatic strategy that may be enacted in concert with larger-scale reductions in greenhouse gas emissions. It is a strategy that is cost-effective, politically tenable, and highly compatible with other elements of sustainable urban growth. In short, it is a no-regrets option for cities concerned about the growing impacts of ambient heat on urban air quality and public health.

Notes

1. Global climate data was obtained from the Goddard Institute for Space Studies (www.giss.nasa.gov).
2. As discussed below, nitrogen oxides and volatile organic compounds constitute the primary chemical precursors to ozone formation.
3. The national Air Quality Index for ozone is a normalized measure of ambient ozone concentrations. An AQI value of 100 is approximately equivalent to the 8-hour national ambient air quality standard for ozone of 80 parts per billion. Ozone AQI values exceeding 100 are considered to be unhealthy for sensitive subpopulations such as children, the elderly, and individuals suffering from respiratory illnesses.
4. It should be noted that when averaged across the nation as a whole, annual ambient ozone concentrations remained stable between the years of 1990 and 1999 (EPA, 2000a). The data reported herein are for the 50 largest U.S. metropolitan regions, listed in Table 1.
5. For the purposes of this study, an ozone “exceedance” is presumed to occur any time the Air Quality Index for ozone equals or exceeds a value of 100.
6. The EPA (1997) has estimated that approximately \$20 billion was spent on Clean Air Act compliance in 1990.
7. As the average residential roof was found to be roughly 200 m² in area, the average surcharge for highly reflective roofing materials was estimated to be less than \$60 per house.
8. Far from recognizing heat as an air pollutant, the EPA’s approach to ozone measurement may serve to mask the role of climate in the ozone production process. As noted in the annual *AirTrends* report:

Because sunlight and heat play a major role in ozone formation, changing weather patterns contribute to yearly differences in ozone concentrations. To better reflect the changes that emissions have on measured air quality concentrations, EPA is able to make analytical adjustments to account for this annual variability in meteorology. (EPA, 2000a, p. 7)

The effect of this adjustment is to decrease reported ozone concentrations for unusually warm years, statistically modifying the empirical ozone record to more closely correspond to trends in precursor emissions. The clear limitation of this approach is that rather than illuminating an actual improvement in urban air quality, it serves to mask a potentially significant threat to urban populations in the form of climate-induced air pollution. The classification of heat as a regulated air pollutant would require the EPA to address heat as a recognized precursor to ozone formation rather than as a meteorological aberration.

9. The reauthorization bill for TEA-21, known as the Safe, Accountable, Flexible, and Efficient Transportation Equity Act (SAFETEA), is, at the time of this writing, under consideration by Congress.

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