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LOCAL POLICY RESPONSES TO URBAN AIR POLLUTION AND ECOSYSTEM STRESS

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OVERVIEW

Urbanization is proceeding rapidly across the globe. During the 20th century, the urban population grew from approximately 200 million to 2.9 billion worldwide [1]. By 2030, the United Nations predicts cities will be home to another 2 billion residents [2]. Such rapid growth is likely to have widespread consequences for urban ecosystems, which in some places are already stressed due to changes in land cover; air pollution; local, regional, and global climate; water quality and availability; and biodiversity. Yet, few studies have focused specifically on the influence of urbanization on key ecosystem services, such as the provision of clean water and air. As a result, the implications of coming urbanization for local sustainability efforts remain underdeveloped.

The goal of this paper is to map ecosystem stress and response strategies with respect to urban air pollution. To this end, the paper briefly summarizes the conceptual linkages between urbanization, air quality, and ecosystem services. The paper next reviews existing research on areas already stressed by air pollution, compiles predictions regarding future urbanization and its likely air quality impacts, and compiles information regarding locally-adopted sustainability strategies to deal with coming air pollution stress. The paper concludes with a summary of current research gaps and an agenda for future research oriented towards local sustainability efforts.

URBAN ECOSYSTEMS AND AIR POLLUTION

Envision an urban ecosystem as a three-dimensional box placed over the earth's surface, containing the materials and agents operating within the city, plus a layer of atmosphere above the earth's surface and a layer of soil below the surface. The box has holes punched on all sides so that it can both receive and expel materials into the broader environment. This city box is nested within other virtual boxes, such as for the biome, continent, and global environments. Each city is also nested

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with the larger network of cities. The status of an urban ecosystem at any point in time is the result of processes that occur within and outside the box.[2]

Urban ecosystems are "human-dominated," meaning that humans are the primary – but not exclusive – agents functioning within the city [4]. As such, urban ecosystems are positioned within the natural (nonhuman) environment and within humans' social environment. Any attempt to understand the functioning of urban ecosystems must include both natural factors and social factors, and their interactions [5]. Some scholars now refer to urban ecosystems as "linked human and ecological systems" [6].

This analysis focuses specifically on the functioning of urban ecosystems with respect to air pollution. In this context, the air quality experienced by city residents at any point in time is the result of both natural and social processes operating within the urban ecosystem.

Air pollution can be produced as a byproduct of natural systems or as a result of human activities (i.e., anthropogenic), such as burning wood for energy that releases hazardous compounds such as particulate matter. Cities may also receive air pollution produced elsewhere and carried into the urban ecosystem by currents in the atmosphere. Likewise, some of the pollution produced within cities may be carried out of the city by air currents. The extent to which air pollution flows into or out of the urban ecosystem depends on its residence time in the atmosphere, meteorology, and physical geography, such as the city's latitude, elevation, topography, proximity to water or mountains, and proximity to other pollution sources.

Other chemical and biophysical processes can reduce the human health impact of air pollution. Small amounts of pollution may be diluted in the atmosphere, limiting their impact. Ecosystems also have some capacity to cleanse the air of harmful pollutants. For instance, some of the pollution may interact with compounds within the city's environment to form new compounds that may be less hazardous for human health. Some pollution particles may merge with water droplets in the air and return to the urban surface when it rains or snows (a process known as wet deposition). Overall, this atmospheric cleansing capacity (elsewhere discussed as air filtration, purification, or assimilative capacity) is a "regulating" service provided by ecosystems [6].

Urbanization places a burden on natural systems to regulate air quality. Urbanization is the human process of developing land in an intensive manner. The more people in the confined space of a city, the more activity that occurs within the city, the more pollution that may be released in that confined space, and the less capacity the ecosystem may have to cleanse the air of harmful pollutants. Even if the city's pollution is blown quickly away, areas downwind from the city may be affected.

At the same time, not all urbanization is the same. Different urban activities and land use intensities can have vastly different implications for air quality. Likewise, urbanization may hasten human social processes to reduce air pollution or to aid the removal of pollution from the urban environment. Residents may, for instance, preserve vegetation that absorbs hazardous air

[2] This approach of representing cities as ecosystems contrasts with other views of cities as home to various ecosystems, such as those found in urban forests, lawns, and water bodies [3].

pollution within or near the urban boundary, or use less polluting fuels for energy. Residents may negotiate with others upwind to reduce the amount of pollution produced outside the city. Some scholars argue that as cities develop economically, their residents become less willing to tolerate pollution and take steps to improve their local environment [7-9].

Given this conceptual foundation, one can imagine that each urban ecosystem likely followed a different path to reach its current state, depending on its unique physical geography, local activities, chemical and biophysical processes, and the highly dynamic interactions of natural and social systems. Each urban ecosystem is also likely to follow its own path into the future. Even with this known heterogeneity, existing research provides some indication as to how urbanization may impact human health and ecosystem service provision, specifically air quality regulation, in the years to come.

The remainder of this paper focuses on two urban air pollutants – ozone and particulate matter – that pose a threat to human health and that are likely to stress the air regulating capacity of urban ecosystems worldwide.

Surface-Level Ozone

Surface-level or tropospheric ozone (O_3) is an air pollutant that affects human health and vegetation and contributes to global climate change. Some ozone migrates towards the earth's surface from the upper atmosphere (stratosphere-troposphere exchange, or STE). The majority of surface-level ozone results from the chemical reactions of carbon monoxide (CO), methane (CH_4), and non-methane volatile organic compounds (NMVOCs) in the presence of nitrogen oxides (NO_X) and sunlight. These ozone "precursors" tend to be produced by industrial, transportation, and agricultural sources, although some natural sources can be significant (such as emissions of NO_X from soils or lightning and VOCs from vegetation).

Temperature and humidity influence the formation of ozone. Surface ozone formation proceeds most efficiently during periods of high temperatures and low humidity. Thus, ozone concentrations experience wide daily and seasonal fluctuations, following meteorological conditions. Extreme ozone concentrations may also arise during high-pressure events when warmer air traps cooler air near the earth's surface, especially in river valleys and near coasts. When combined with low wind speeds, these temperature inversions can trap pollution for days, building up to levels that are severely dangerous for human health.

Ozone and its precursors often migrate away from sources, potentially resulting in broad downstream impacts from local pollution. For instance, CH_4 and CO are long-lived and may travel widely from their sources (over 1,000,000 km and over 25,000 km, respectively), while NO_X react quickly with other compounds and thus have a much more limited travel range (about 200 km) [10]. Thus, urban ecosystems may experience elevated ozone levels because of local emissions of ozone precursors and because of elevated "background" levels of ozone from long-lived precursor emissions produced elsewhere.

Despite the substantial amount of ozone chemically produced each year, ozone is highly reactive with other compounds, resulting in large-scale loss of much of the ozone produced each year. Ozone also reacts with plant, soil, and built surfaces, in a process called deposition, reducing surface-level ozone concentrations but potentially adversely affecting vegetative health.

In addition, the oxygen molecules in ozone frequently disassociate in the presence of UV sunlight and then recombine into ozone, but some portion of the disassociated oxygen contributes to the formation of hydroxyl radicals (OH), especially in high humidity [11]. These hydroxyl radicals then combine with other reactive carbon and nitrogen compounds, such as methane and carbon monoxide, forming less harmful compounds and contributing to the atmosphere's "self-cleansing capacity" [11]. The self-cleansing cycle is largely perpetuated by NO_X, which contribute to ozone formation and the recycling of OH. In low NO_X (or NO_X-limited) environments, ozone is destroyed and OH concentrations decline. Thus, NO_X emissions contribute to ozone formation but also increase the availability of OH, which keeps down other pollutant concentrations and can keep net ozone production in check [11]. Some evidence suggests that global OH concentrations have declined by about 10% over the past few centuries [6].

Whether NO_X emissions contribute to net ozone production, especially in urban areas, appears to depend on the complex relationship between NO_X and VOCs [12]. In locations with low NO_X levels, ozone formation proceeds rapidly with increased NO_X and shows little sensitivity to increased VOCs (known as NO_X-sensitive chemistry). In areas with already high concentrations of NO_X, ozone increases with VOC emissions and may decrease with higher NO_X emissions (known as VOC-sensitive or NO_X-saturated chemistry) [12]. These chemical interactions indicate that pollution control policies may yield different results depending on whether the area is NO_X-sensitive or VOC-sensitive. In particular, VOC-sensitive areas may see ozone concentrations increase with reductions in NO_X emissions [12]. Some evidence suggests that urban centers tend to be VOC-sensitive, while more rural areas tend to be NO_X-sensitive, although more research is needed to identify the areas subject to NO_X- or VOC- sensitive chemistry [12].

Past and Present Concentrations

Complexity in the biochemistry and geography of ozone production and removal makes ozone pollution at any particular location highly dependent on latitude, elevation, meteorological conditions, local and upstream emissions of precursors, and the presence of other reactive compounds. Table 1 and Figure 1 illustrate the range of ozone concentrations recently observed in selected cities: from 100 $\mu g/m^3$ in Osaka to 600 $\mu g/m^3$ in Mexico City [13]. High concentrations in Mexico City, Santiago, and Los Angeles tend to occur during extreme temperature inversions.

The lack of widespread, long-term, and internationally comparable monitoring efforts makes evaluating ozone concentrations, trends, and exposure difficult for many cities worldwide. To facilitate policy-making, atmospheric chemistry transport models (CTMs) are often used to estimate concentrations as well as to project concentrations under different scenarios. Different models can yield substantially different results, however, depending on model complexity and inputs (typically national-level emissions inventories and meteorological data). A complete discussion of available models and results is outside the scope of this paper. The following discussion highlights a few of the recent model results.

A simulation using the MOZART-2 CTM found that the global ozone burden resulting from human activities increased approximately 50% from 1860 to 2000 (254 Tg to 372 Tg), with the majority of this increase since 1950 [14]. The simulation further determined that the global average surface ozone concentration (seasonally averaged, daily 1-hour maximum, population-weighted) resulting from human activity increased from about 20 parts per billion (ppb) in 1860 to nearly 60 ppb in 2000 [15]. Ozone concentrations were estimated to be highest in 1860 in parts of Asia and South America, and highest in parts of Asia and North America in 2000, with some places experiencing concentrations in 2000 more than 60 ppb over pre-industrial levels [15].

The concentration estimates can be combined with epidemiological research to estimate the human health impact of anthropogenic ozone [13]. While extreme ozone concentrations are cause for immediate concern over public health, ozone can trigger health impacts at even low levels and the bulk of its impact is expected to come from chronic exposure to lower concentrations [16]. Anenberg et al. found that ozone pollution in 2000 resulted in an estimated 0.7 million (± 0.3 million) respiratory mortalities globally, with 75% of these deaths estimated in densely-settled Asia [15]. While such estimates are a useful benchmark for estimating human health impact, the results are highly uncertain especially for urban areas due to the coarse spatial resolution of the atmospheric model (here, 2.8° latitude by 2.8° longitude) [15].

Another well-known simulation effort employed the TM3 CTM (spatial resolution of 10° longitude by 7.5° latitude) with emissions data from the global RAINS model [17, 18]. The simulation estimated a global ozone burden of 462 Tg in 2000, with the highest relative burden from the mid-latitudes of the Northern Hemisphere. The simulation estimated that global average ozone concentrations during 1990-2000 were 30-60 ppb [17]. The highest estimated average concentrations were found in the Northern Hemisphere, especially off the Eastern U.S., in the Mediterranean, and in Central and Eastern Asia. The authors noted that stratospheric transfer accounted for high concentrations at high elevations in the Himalayas. The TM3 simulation may have underestimated concentrations from biomass burning in the Southern Hemisphere (especially in Africa) as compared to other CTMs, such as STOCHEM [17].

The TM3 modeled concentrations have also been used to estimate ozone exposure specifically in world cities for the OECD [10]. The analysis applied only to cities with populations of at least 100,000 in 2000, which together housed approximately 2 billion urban residents. The model calculated exposure based on the average concentrations experienced within model grid cells covering the cities. The model found that urban ozone concentrations in 2000 were highest in Asia (especially India, Korea, South Asia, and Japan), the Middle East, and North Africa, with concentrations in these locations averaging at least 15 ppb over a 35 ppb health threshold [10]. Elevated concentrations were estimated to cause approximately 19,000 premature mortalities a year worldwide, which translates to about 7 mortalities per 1 million urbanites. When accounting for the size of their urban populations, premature mortalities from ozone exposure were estimated to be highest in North America, Russia, and Asia (at 16, 15, and 14 mortalities per 1 million urban inhabitants, respectively) [10].

The predicted human health impacts from urban ozone exposure reported in the OECD study [10] were substantially smaller than what was found more recently by Anenberg et al. [15] for the entire global population. The difference is likely due to consideration of only larger cities, to consideration of health impacts only for exposure to levels of 35 ppb and higher, and to the coarser resolution of the TM3 model used in the OECD study, as well as to other differences in model complexity and inputs between the two studies.

Model resolution is often mentioned as a major drawback to using global CTMs for estimating urban air quality [16, 19]. Cape argued that "the spatial scale of global models, particularly the vertical resolution, is still sufficiently coarse that accurate predictions of O_3 exposure cannot yet be made, even for present-day conditions, at spatial scales relevant to predicting effects at a specific site" [20]. Sillman also argued that global CTMs have substantial uncertainties regarding effects of NO_X and VOC on ozone formation, and suggested that additional efforts are needed to validate present-day model results with direct measurements [12].

Scenarios and Projections

Given model variation and uncertainties, researchers typically warn that it is difficult to predict future ozone concentrations. Future levels depend on: (1) anthropogenic emissions of ozone precursors; (2) emissions from vegetation, especially of VOCs; (3) the contribution of ozone from the stratosphere (STE); and (4) the efficiency of the ozone formation and loss processes [20]. All major processes are subject to their own uncertainties, and will also be influenced by future climate variability. For example, higher temperatures and more cloud-free days will improve the efficiency of ozone production, while plants experiencing drought-stress will be less efficient at absorbing ozone via dry deposition and will release more biogenic emissions, contributing further to ozone production [16]. In addition, both methane and ozone are potent greenhouse gases [17, 21]. Thus, to accurately predict future ozone concentrations, researchers need to have a good understanding of future emissions scenarios, which depend in large part on demographic, economic, and land use changes, as well as likely climate and meteorological changes, and climate interactions with the other sources of change. Most models can incorporate only a limited range of possibilities, and thus can provide only a limited range of estimates about future air quality.

Many simulation studies use scenarios developed for the Special Report on Emission Scenarios (SRES) by the Intergovernmental Panel on Climate Change (IPCC). None of the six SRES scenario classes specifically modeled changes in urbanization patterns worldwide. Instead, the scenarios differ in their population and economic growth patterns, in energy demand, in use of technology, and in land use (see Table 2). The B2 scenario comes the closest to following the UN's current global population growth estimates, which anticipate approximately 9.1 billion residents worldwide by 2050 [2]. The A1 and B1 scenarios project slightly slower population growth than the UN estimates, while the A2 scenario project significantly faster population growth, with a global 2050 population estimated at around 11.3 billion.

Anthropogenic emissions were calculated for the IPCC SRES scenarios using many different models, each with different model complexity and input data, to bound the range of plausible emissions estimates [22]. For simplicity, this analysis discusses only the emissions estimates produced by the

IMAGE modeling team.[3] Global emissions of NO_X, NMVOC, CO and CH_4 were all projected to be highest under the A2 scenario and lowest under the B1 scenario by the end of the century. Emissions of NO_X and CH_4 were projected to peak at or before 2050 under the other four scenarios, while NMVOC emissions were projected to peak closer to 2030. Carbon monoxide emissions were projected to be stable or decline under all scenarios, owning to changes in heating fuels and vehicle technology.

Atmospheric models were then used to translate the projected precursor emissions under the SRES scenarios into estimates of the global ozone burden. The MOZART-2 simulations, for instance, found that the global ozone burden could increase by another 40% between 2000 and 2100 under the IPCC's A1FI and A2 scenarios [14]. Under the extreme A2 scenario, the ozone burden is projected to be highest in South and East Asia, North Africa, the Middle East, and the tropical South Atlantic by 2100 [14]. The MOZART-2 simulations also found that the global ozone burden would increase modestly (by 12%) under the A1B scenario and decrease slightly (-6%) under the B1 scenario by 2100 [14]. In all scenarios, the simulated ozone burden from MOZART-2 trends track closely with the projected emissions for NO_X [14].

Various modeling teams also estimated the change in surface ozone concentrations likely to result from the SRES emission scenarios (using CTMs with a spatial resolution of 5˚ latitude by 5˚ longitude). The extreme A2 scenario was estimated to produce increases in average ozone concentrations by up to 25 ppb in mid-latitudes of the Northern Hemisphere and by 7-16 ppb elsewhere from 2000-2100 [19]. Other model scenarios predict smaller changes of up to 10 ppb on average, although the high-efficiency scenario (B1) predicting reduced ozone concentrations of 4 ppb on average by 2100 [19]. By 2030, most scenarios predicted that ozone concentrations may rise by an average of 5 ppb (ranging from 2 to 7 ppb) in the Northern Hemisphere [19]. According to these models, emissions of CH_4 accounted for approximately half of the predicted change in near-term ozone concentrations, while NO_X emissions accounted for most of the rest of the predicted change [19].

More recent efforts have updated the SRES emissions scenarios to account for recent national commitments to control emissions [17, 18]. The projected emissions under current legislation were then employed to estimate the increased global ozone burden and concentrations resulting from precursor emissions [17]. The TM3 model projected an increased global ozone burden of 8% from 1990 to 2030 (450 Tg to 485 Tg) under the current legislation scenario, and a reduction of 4% by 2030 (to 430 Tg) when applying the maximum feasible technology [17]. Such an increase in the global ozone burden under current legislation was estimated to result in increased concentrations of 4-6 ppb during 2020-2030 as opposed to 1990-2000 over the Atlantic Ocean and North Pacific, and increased concentrations of 8-12 ppb in South and East Asia (especially India, Pakistan, Bangladesh, and China) [17]. These estimates narrow the projected near-term growth in ozone concentrations over what was found using the earlier IPCC SRES scenarios. Note that the

[3] For information on different models used with the SRES scenarios, see http://sres.ciesin.org/OpenProcess/htmls/Model_Descriptions.html. See also http://www.chem.uu.nl/nws/www/publica/Publicaties2005/E2005-269.pdf for evolution of IMAGE model and its application in global climate studies.

projections were for average ozone concentrations, indicating that some locations may experience maximum concentrations quite a bit higher than the projected concentrations.

The TM3 simulation revealed that ozone concentrations were likely to increase in Europe and North America by 2030 despite projected decreased local precursor emissions [17]. Two processes may be at work here. First, Europe and North America may experience some increased ozone because of local reductions in NO_X emissions, which had been keeping ozone concentrations lower than otherwise through titration and because these areas may already be NO_X-saturated [16]. Second, these areas are likely to be impacted by intercontinental transport of ozone and precursor emissions especially from South and East Asia, where southern latitudes improve the efficiency of ozone formation and where atmospheric currents effectively lift pollutants into the upper atmosphere to be transported long distances [16, 23]. One study found that Asian emissions may be responsible for increased spring background ozone concentrations in the Western U.S. of approximately 10 ppb between 1984 and 2002 [24]. Likewise, emissions from North America were estimated to increase surface ozone concentrations in Europe by 2-4 ppb, on average, and 5-10 ppb during specialized events [25].

The TM3 simulation confirmed that future CH_4 pollution controls could have a strong impact on future ozone concentrations [17]. Methane has a long atmospheric lifetime and is not very reactive, and thus can circulate widely from its emission source and contribute to increased background levels of ozone throughout the world. One study estimated that “a 20% increase in the [global] methane burden could lead to a 2-5% increase in the [global] ozone burden” [16, 22]. Methane is converted to ozone (a process known as methane oxidation) most efficiently in warm climates in low latitudes and in the presence of NO_X [16]. Limiting NO_X emissions, especially in NO_X-sensitive areas in the low and mid latitudes, could limit methane oxidation and its contribution to background ozone, as would reducing methane emissions altogether.

Another recent modeling effort extracted the ozone concentrations predicted for the CLE scenario for 2000 and 2030 [17] to estimate average ozone concentrations specifically in world cities [10]. The modeling team then calculated the potential exposure of the urban population to average ozone levels greater than 35 ppb, and used the estimated exposures to calculate health impacts from elevated average ozone concentrations [10]. The analysis revealed rapid growth in potential elevated ozone exposure in cities in India and South Asia from 2000 to 2030 (see Figure 6) [10]. Despite slower growth rates, potential exposure levels were projected to also be high in 2030 in cities in Japan, Korea, the Middle East, and North Africa [10]. The modeling team estimated that ozone exposure could result in 88 excess mortalities per million urbanities in Asia by 2030, well above the estimated 57 mortalities per million urbanities in North America [10]. Worldwide, urban exposure to elevated ozone levels in 2030 was predicted to result in approximately 150,500 excess mortalities [10].

Potential climate variability adds substantial uncertainties to ozone projection efforts. Most air quality simulation models use past meteorological data combined with projected emissions changes at country or broad regional levels to estimate future air quality. These models include few, if any, meteorological impacts and feedbacks from possible climate changes [19, 20]. In addition, few models incorporate spatially-explicit land use or forestry changes likely to result from development pressures or from climate change, which may influence emissions rates from vegetation as well as capacity of the environment to adapt to air pollution stress [16, 20]. Emissions of VOCs, in particular, from vegetation can be quite high and can influence the NO_X-VOC chemistry within urban regions.

To improve projections given explicit land use and climate scenarios, finer-scale regional climate models (RCMs) may be used [16]. For instance, one European RCM simulation projected increased ozone concentrations in southern and central Europe due to decreased precipitation and cloudiness resulting from climate change (which promote ozone production efficiency) and from an increase in biogenic VOC emissions from vegetation under climate stress [16]. The models also projected decreased ozone concentrations in northern Europe due to wetter and cloudier conditions likely from climate change [16]. Even with the finer spatial resolution, RCMs may still not be appropriate to estimating air quality within particular cities, especially during extreme events.

Particulate Matter

In addition to facing health impacts from elevated ozone concentrations, many cities worldwide also suffer from elevated concentrations of particulate matter (PM). PM is typically produced during the combustion of fossil fuels and can include a wide variety of substances suspended in the air that can become lodged in the lungs during respiration. Researchers often focus on particles with a diameter less than 10 µm, termed PM_{10}. Recent focus has shifted specifically to fine particulates with a diameter less than 2.5 µm, termed $PM_{2.5}$. Some PM is emitted directly during combustion activities (such as black carbon and organic carbon), while other PM is formed from precursors NO_X, sulfur dioxide (SO_2), and ammonia. PM is also produced by natural activities, such as volcanoes, dust storms, forest fires, and sea spray.

PM tends to have a shorter atmospheric lifetime than ozone, typically remaining in the atmosphere for only a few days and traveling up to 800 km from sources [10]. The shorter travel distances for PM means that elevated concentrations tend to be more localized than ozone, exhibiting steep gradients around source activities.

Past and Present Concentrations

Figure 1 presents the range of PM_{10} concentrations recently observed in selected cities worldwide: from 20 µg/m^3 to 220 µg/m^3. The highest concentrations are typically found in Asia, resulting from a combination of fires, dust storms, industrial activity, and use of polluting heating fuels, such as coal [13]. Average annual PM_{10} concentrations in Karachi, for instance, recently reached 220 µg/m^3. Average annual PM_{10} concentrations above 100 µg/m^3 have also been recently observed in New Delhi, Kathmandu, Dhaka, Calcutta, Shanghai, Beijing, Gangzhou, Cairo, Lima, and Arequipa [13]. The World Health Organization (WHO) recommends average annual PM_{10} concentrations of only 20 µg/m^3 to protect human health, with interim targets of 70, 50, and 30 µg/m^3 [10].

While observational data are more widely available for PM_{10} in cities than for ozone, the networks remain sparse in many developing countries. For this reason, the World Bank produced econometric models to estimate concentrations of PM_{10} in more than 3,000 cities worldwide based on available measurements and on the cities' and their country's population, demographics, development level, energy fuels used, and meteorology [26]. The latest city-level data from 2006 revealed wide disparities in the estimated average annual concentrations of PM_{10} (see Figure 7). Many cities in North and Central Africa, the Middle East, and Asia were estimated to have PM_{10} concentrations in 2006 well above even the WHO's highest interim target of 70 μg/m^3. The most polluted cities in 2006 were located in Sudan, Nigeria, and Pakistan, with estimated PM_{10} concentrations above 200 μg/m^3, as has been observed in Karachi. The World Bank data also revealed that average urban PM_{10} concentrations have declined since 1990 in most all countries worldwide, likely due to economic development, technology improvements, and pollution controls such as for SO_2.

Observational data are even less available for $PM_{2.5}$, despite the recent shift in focus towards smaller particles. For this reason, researchers have employed atmospheric models to estimate $PM_{2.5}$ levels as has been done for ozone. Simulations indicate that the global burden of anthropogenic $PM_{2.5}$ pollution has also increased significantly over pre-industrial levels. One study using the MOZART-2 CTM estimated that the global annual average maximum $PM_{2.5}$ concentration (population-weighted) increased from 1.1 μg/m^3 in 1860 to 16.1 μg/m^3 in 2000 [15]. Concentrations were estimated to be highest in parts of North and South America in 1860, and highest in parts of Asia in 2000 [15]. Modeled concentrations of fine particulates in 2000 were estimated to cause 3.5 million (± 0.9 million) cardiopulmonary mortalities and 220,000 (± 80,000) lung cancer mortalities per year worldwide [15]. The estimated mortalities from fine particulates were concentrated in Asia (75%) and Europe (17%) [15].

Recent analysis of $PM_{2.5}$ with satellite-derived concentration estimates at a fine resolution (0.1˚ latitude by 0.1˚ longitude) yielded a global population-weighted geometric-mean concentration estimate during 2001-06 of approximately 20 μg/m^3 [27].[4] Eastern Asia was estimated to have the highest geometric-mean concentrations (of 34 μg/m^3), with levels over 100 μg/m^3 in the cities of eastern China and 80-100 μg/m^3 in India. The authors confirmed that many of the highest concentrations were in urban and industrial areas. The analysis estimated that half of the population of Eastern Asia was exposed to average $PM_{2.5}$ concentrations above 35 μg/m^3, as was 38 percent of the population in Central Asia and 17 percent in North Africa [27]. The estimates were validated using monitored data in the United States, Canada, and Europe, but could not be validated for much of the rest of the world due to sparse monitoring facilities [27].

Scenarios and Projections

The World Bank's econometric models were used to estimate PM_{10} concentrations in world cities under current legislation for 2030, as was done for ozone (see Figure 8) [10]. The analysis revealed that cities in Asia (except Japan and Korea), the Middle East, and Africa could all experience average annual PM_{10} concentrations substantially higher than the WHO's 70 μg/m^3 interim target by 2030, affecting more than 80% of the urban population in those regions [10]. By comparison, only about

[4] Similar satellite-derived concentration estimates on finer resolution grids are being constructed for ozone (at 0.5˚ by 0.5˚) and for nitrogen dioxide (at 0.1˚ by 0.1˚), which can be used to more accurately gauge urban air pollution (personal communications with Valentina Mara at Columbia University, August 20, 2009 and Erica Zell at Battelle Memorial Institute, March 25, 2010).

20 percent of the urban population in Eastern Europe and Latin America (excluding Brazil) were estimated to experience such high levels by 2030, with less than 5% in other regions [10]. Urban PM_{10} concentrations were estimated to substantially increase mortalities, especially in China and South Asia, resulting in nearly 1.9 million premature deaths per year worldwide by 2030 (see Figure 9) [10]. Even rapid implementation of best available technology was unlikely to reduce concentrations below the WHO's 70 $\mu g/m^3$ interim target for many of the most polluted cities in Asia, the Middle East, and Africa by 2030 [10].

IMPLICATIONS FOR URBAN SUSTAINABILITY EFFORTS

The recent air quality monitoring and modeling literature reveals several implications for cities and efforts towards urban sustainability.

First, the air quality impact of coming urbanization is difficult to forecast, given the variety of factors influencing local air quality and the complex atmospheric chemistry of pollution production and removal. Climate change adds considerably to the complexity of forecasting air pollution, as changing weather patterns will influence the atmospheric lifetime and spatial dispersion of pollution away from sources. Drier and hotter weather conditions may result in broader impacts of local pollution on downwind locations, while wetter weather may narrow impacts on downwind locations. Changing weather patterns may also influence the production or removal of pollution by vegetation, and the production of PM by natural sources such as dust storms and forest fires. Thus, even without other changes in urbanization rates or emissions, urban air quality is likely to change in coming decades, making some places better off and some places worse off than before.

Second, the majority of the population exposed to elevated pollution levels in the future will live in cities. If UN projections prove accurate, another 2 billion residents will be living in urban agglomerations by 2030, bringing the world's urban population to nearly 60% of the total [2]. Both ozone and PM concentrations already are highest in urban and industrial areas. Rapid urban and economic growth coupled with already high population densities and pollution concentrations, especially in Asia, the Middle East, and North Africa, will likely pose serious health concerns for the world's urbanites in coming decades. The situation may be most severe for the world's poorest cities, which may have few monetary resources to combat local air pollution [13].

Third, economic production and local consumption activity within cities will likely continue to be an important source of air pollution globally. An analysis of 2005 emissions from the EDGAR emissions inventory revealed that approximately half of the global anthropogenic emissions of NO_X, VOCs, and CO and a third of the global SO_2 emissions were produced within cities [28]. The largest urban agglomerations emitted disproportionately large amounts of pollution, such as from the Tokyo-Nagoya-Osaka corridor and from the New York-New Jersey-Philadelphia corridor. Pollution produced within the world's megacities may exert a substantial impact on downwind locations, such as in Western U.S. and in Western Europe [29].

Only a few of the most populated cities worldwide also had extremely high emissions densities when viewed in terms of their associated land area [28]. The largest cities with high densities include Lagos and Shanghai for CO, and Zhenjiang, China for NO_X and SO_2. Instead, the highest pollution densities were typically found in small to mid-sized cities in Asia, Africa, and Eastern Europe. High pollution densities may translate into poor local air quality for cities, especially in regions with weak atmospheric cleansing capacity [30].

Fourth, cities may also suffer elevated pollution concentrations because of emissions produced elsewhere, compounding local emissions and potentially countervailing local pollution control efforts. For instance, the simulation models revealed increased ozone concentrations in North America and Europe in coming decades despite decreased local emissions in these areas. The simulation models also revealed the potentially large impact that poorly-controlled methane emissions have on ozone concentrations worldwide. Global cities emitted only one quarter of the anthropogenic methane emissions in 2005, and highly developed cities produced very few methane emissions [28]. Thus, cities might instead work to reduce the conversion of methane to ozone within cities by reducing local NO_X emissions. Similarly, emerging research into the complex NO_X-VOC chemistry suggests that cities in NO_X-sensitive regions may need to pursue different strategies than cities in VOC-sensitive regions in order to reduce local ozone concentrations.

Local Strategies for Adaptation and Mitigation

Much of the policy dialogue on reducing urban air pollution has historically surrounded efforts taken at the national and international scales. The United Nations, the Organisation for Economic Co-Operation and Development, the World Bank, and other regional organizations such as the European Commission have all been key players in international discussions of air quality policy and management. One of the predominant forums for establishing international air quality policy has been the UN's Geneva Convention on Long-Range Transboundary Air Pollution (CLRTAP), established in 1979, with its eight protocols covering sulfur and nitrogen oxides, VOCs, ozone, heavy metals, and persistent organic pollutants. Fifty-one countries in Europe, Asia, and North America have since ratified CLRTAP. Other international treaties have covered stratospheric ozone depletion, production and transport of hazardous waste, and prevention of catastrophic climate change, among other topics.[5]

While national representatives have been proposing and negotiating future air quality policy, some of the world's largest cities have individually been at the forefront of the fight against air pollution. The following discussion highlights some common strategies for reducing local emissions and maintaining air quality in the face of rapid urbanization. The discussion includes examples of cities where these strategies have recently been employed, and where available, notes the expected or actual impacts of policy efforts on urban air pollution. In addition to the policies mentioned here, several "no regrets" air quality policies for cities have been advanced and are included at Figure 10 [31]. In general, successful policy strategies must be tailored to local conditions, including local emissions, regional pollution levels, local geography and meteorology, likely impacts from climate change, stage of economic development, and available administrative and financial resources.

[5] For a list of all the UN's environment treaties, see http://treaties.un.org/pages/Treaties.aspx?id=27&subid=A&lang=en.

POLLUTION FROM VEHICLES

Vehicle emissions are a primary source of air pollution in many cities worldwide and are often responsible for much of the recent growth in urban air pollution. City-level policy efforts to combat vehicle pollution often focus on reducing NO_X and PM emissions from diesel fuel engines. For instance, European cities adopted a continent-wide standard for diesel emissions known as EURO I-IV. The standard has become more stringent over time, such that a EURO IV bus emits a third less NO_X than a EURO III bus [32]. Since 2006, Paris has required all trucks and utility vehicles to meet EURO IV standards [33]. In 2010, the Greater London Authority passed legislation requiring all city buses to meet EURO IV standards for NO_X and PM by 2015 [32].

Cities outside Europe have also adopted the EURO standards, such as Hong Kong, which provides grants to commercial vehicle owners who scrap their diesel vehicles and replace them with vehicles in compliance with EURO IV standards [34]. Moscow is putting forward legislation that would adjust vehicle personal property tax amounts depending on each vehicle's emissions standards [35]. Vehicles that conform to the EURO IV standard would have their tax assessments unchanged, while vehicles with higher emissions would see their assessment increase. In 2010, Moscow aligned itself with the rest of Europe by mandating all new vehicles meet EURO IV standards.

Malaysian cities have been proactive in the fight against air pollution since the landmark Motor Vehicle Control of Smoke and Gas Emissions ordinance of 1977. As a result, cities like Kuala Lumpur have developed an elaborate maintenance and inspection system, known as the Area Watch and Inspection program, in which vehicles are subject to strict inspections and heavy penalties are levied upon those who do not meet the latest standards [36]. In 2009, Kuala Lumpur adopted the EURO IV standard.

Tokyo has taken an aggressive approach by mandating that no diesel vehicles be driven, sold, or bought in Tokyo [37]. Since the beginning of 2010, this policy has led to the replacement or disposal of over 149,000 vehicles. The remaining diesel vehicle fleet that has not been scrapped or replaced is required to install exhaust purifying devices, and purchase fuel from approved light-oil low-sulfur petroleum stations. Several other cities now require the installation of exhaust purifying devices on commercial diesel vehicles. In 2005, New York City passed local laws 39-42 mandating the use of low-sulfur fuel and emissions control technology on all city motor vehicles, vehicles used in the fulfillment of city contracts, sight-seeing buses, vehicles that transport children to and from school, and off-road vehicles used in city construction projects [38]. Boston enacted a similar regulation in 2007, demanding the installation of emissions-control equipment (retrofits) on all on- and off-road diesel vehicles [39].

Since 2002, the Indian city of Pune and the U.S. Environmental Protection Agency (EPA) have been working together on an Air Quality Management Initiative to reduce air pollution. The project calls for increased air quality monitoring, diesel vehicle retrofitting with emissions control devices, and implementation of new technologies that will dramatically reduce PM emissions [40]. A major challenge is the unavailability of low-sulfur diesel fuel in India, which is integral to any emissions reduction efforts. If successful, the Indian government hopes to expand the program to other cities around the country, and has already duplicated the diesel retrofit program in Mumbai. EPA has similar diesel retrofit partnership programs in Mexico City, Bangkok, Santiago, and Beijing. The partnership between EPA and the Chilean capital of Santiago has been effective in reducing CO_2 and PM emissions in Santiago. The policy, initiated in 1999, calls for the replacement of residential wood, kerosene, and diesel boilers with natural gas boilers. The plan also retrofits old buses and taxis with compressed natural gas (CNG) conversion kits and installs diesel particulate traps. Some old buses are also being replaced with new CNG and hybrid diesel-electric buses. EPA estimates that the initiative will reduce residential PM emissions by between 61% and 95%, and public transportation PM emissions by 20% over the next 20 years [41].

As in Santiago, many cities encourage replacement of older high-emissions vehicles with newer, lower-emissions vehicles. Beginning in 2012, London will enforce a 15-year rolling age limit for taxis and a 10-year rolling age limit for all other private-hire vehicles [32]. Since 1999, the city of Austin, Texas, USA has implemented a program designed to replace the city-owned fleet by purchasing liquefied petroleum gas (LPG), CNG, electric, and hybrid vehicles [42]. As a result, Austin has increased alternative fuel use by 25%. Hong Kong provides grants to taxis and light buses that operate LPG vehicles. These grants, in concert with strict emissions standards, have reduced street-level PM and NO_X concentrations in Hong Kong by 33% and 31%, respectively, in the last 10 years [43].

In 2001, the Delhi city government began the replacement of diesel fueled commercial vehicles with CNG vehicles [44]. A study conducted in 2006 to determine the impact of the policy on air quality found notable decreases in polycyclic aromatic hydrocarbons (PAH), SO_2 and CO concentrations, but no reduction in NO_X and PM emissions [44]. In 2009, the city of Jerusalem partnered with a private electric vehicle (EV) manufacturer to fund the installation of 100 EV stations within the city by 2011 [45]. The city hopes that providing citizens with a widespread EV infrastructure will reduce emissions from gasoline- or diesel- fueled vehicles. Other vehicle replacement programs have been adopted in Tehran and Cairo [44].

Some cities have restricted vehicles entirely from parts of the city to reduce pollution and population exposure. Mexico City's renowned "Hoy No Circula" program began in 1989 to address severe air quality problems. Under the rule, vehicle owners cannot operate their car 1 day per week, with the day determined by the last digit of their license plate number [46]. Furthermore, vehicles are required to pass emissions inspections twice a year in Mexico City. Inspections were adopted as a policy response to perceived gaming of the system by residents who bought second and often more polluting vehicles [31, 47]. In 2008, the city of Berlin started a sticker rating system for all vehicles within the Berlin central district [48]. The color of the sticker depends on the vehicle emissions and whether it follows Euro I-IV standards, has particle filters, has a 3-way catalytic converter, and has a diesel or gasoline engine. Different color stickers grant or deny the vehicle access to certain parts of the zone. Berlin and Mexico City set themselves apart from most other cities by restricting emissions from all vehicles, not just commercial or city-owned fleets.

Other cities have adopted policies limiting vehicles from their central cities. In 1999, the city of Bangkok enacted a comprehensive air pollution reduction plan that restricts cars and single-occupancy vehicles from certain parts of the city [49]. Santiago now restricts 20% of its vehicles from entering the city on weekdays when the air quality is very poor, and Bogota and Sao Paulo limit vehicles during peak hours [44].

Restricting vehicle access to the city improves traffic congestion as well as limits idling, which limits air pollution. Many cities have taken steps to curb traffic congestion by promoting public transit, flexible work schedules (such as Austin, Texas), and congestion relief projects (such as Boston's "Big Dig"), with varying results. In 2003, the city of London implemented the London Congestion Charging Scheme (CCS), using a single charge of £5 for vehicles entering a central London zone during weekday business hours [44]. A review of the impact of the CCS in 2005 found mixed results. While congestion was reduced, vehicle speed increased due to the lower congestion, and additional buses were required to meet the excess demand for travel in central London [50].

In response to growing concern over the effect of traffic congestion on air quality, Chinese and Indian cities have invested heavily in their public transportation systems. In the last decade, Beijing, Shanghai, Guangzhou, Tianjin, Shenzhen, and Nanjing have built extensive metro systems, and ten more cities have plans to build metro lines in the near future [51]. According to the National Bureau of Statistics, the number of buses in Beijing has increased four-fold in the last twenty years. In order to curb highway congestion, a number of Chinese cities have also adopted light rail systems, which are expensive and have to be subsidized by the government. The bus rapid transit system in Ahmedabad in India, which began its first implementation phase in 2009, will significantly expand the city's bus system by dedicating bus corridors free of traffic, installing numerous and efficient loading stations, and connecting the bus stops to the existing rail metro network. In addition to the expansion of bus lines, the city of Ahmedabad will add two metro lines. As a result, the city won the 2010 Sustainable Transport Award [52].

POLLUTION FROM STATIONARY SOURCES

Many developed cities have long been subject to state, national, or regional industrial pollution control policies aimed especially at reducing SO_2, VOCs, and NO_X. Californian cities have often been at the forefront of industrial air pollution reduction. For instance, the greater Los Angeles region implemented a multi-industry cap-and-trade program in 1993 that sets caps on facility emissions with declining balances of allowable maximum emissions. According to the South Coast Air Quality Management District, "the program has resulted in an additional 68 percent (27,643 tons) and 59 percent (6,073 tons) decrease in allowable emissions for NO_X and SO_X, respectively; and a 62 percent (15,758 tons) and 50 percent (3,611 tons) reduction in actual emissions for NO_X and SO_X" from 1993 to 2007 [53, EX-1].

In 1997, Santiago began implementation of an industrial emissions trading program, in which the most polluting industrial sites were given emissions allowances to trade. In addition to the cap-and-trade component, the plants that pollute the most are forced to shut down on days when the city decrees the air quality to be too poor. As a result, a number of plants switched to natural gas powering, yielding a 40% decrease in PM concentrations and an 80% decrease in potential PM emissions by 2008 [54].

In recent years, cities have also focused on reducing PM emissions from construction or industrial sites. In 2008, San Francisco passed legislation to limit direct PM emissions from open burning of agricultural and non-agricultural waste, emissions from various combustion sources such as cement kilns and furnaces, and dust emissions from earthmoving and construction or demolition operations [55]. The regulation also requires the reduction of indirect NO_X and SO_2 pollution from power plants, industrial facilities, and other combustion sources, and VOCs from petroleum refineries, coatings and solvents, product manufacturing, fuel storage, transfer and dispensing activities, and many other industrial and commercial facilities [55]. New York City requires that any construction site producing dust must have its construction materials wetted with appropriate spraying agents, and that trucks leaving the site cannot be covered with PM [38]. Rule 1155 (2009) in the greater Los Angeles region prohibits the release of visible PM emissions from PM pollution control devices, including bag houses, electrostatic precipitators, scrubbers and other dust collecting equipment [56]. In addition, large PM-emitting facilities must begin weekly monitoring for visible emissions by trained personnel.

Some cities have promoted energy efficiency and conservation as another strategy for reducing pollution. From 1997 to 2005, New York City invested in the conversion of its traffic lights to energy-saving light-emitting diodes, and replaced all school boiler systems from coal-burning to natural gas [57]. In 2006, the city of Paris required its residents' water heaters to have the latest low NO_X burners [33]. In 1999, the city of Austin began to reduce the impact of new growth in 1999 by encouraging urban infill, walkable districts, mixed-use, and transit-oriented neighborhoods. In addition to reducing travel demand and associated emissions, Austin provides financial incentives for green building designs and maintenance techniques that reduce energy and water consumption, helping to reduce emissions [42]. Careful urban and regional planning like in Austin is argued to be a critical response strategy for preventing air quality deterioration in rapidly growing cities, especially in developing nations [31, 47].

Finally, several conclusions have been offered about what is needed to be successful in urban air quality management [31, 47]. First, cities need the technical capacity to monitor and model air pollution, and to identify appropriate solutions for local problems. Cities then need the administrative and legal capacity to implement and enforce air quality policies. In general, capacity is tied to financial resources, posing distinct challenges for less developed cities. This point is also related to having an appropriate regional governance scheme, which is quite difficult in larger cities and urban regions with multiple government actors and sometimes multiple sovereign nations. Cities also need public support for their air quality policies to ensure compliance, which may stem from good communication with residents and the business community. Environmental policies often play second fiddle to economic development policies when perceived to be in conflict, and a persuasive education and informational campaign may be needed to generate the public support and political will to make air quality commitments.

Thoughts on Data Needs and Future Research

This paper examined existing research that generally addresses the relationship between urbanization and air quality on a global scale. So far, the comparative research specifically examining urbanization as a driver of air pollution change, as opposed to other factors like population change, is quite limited.

One of the primary limitations to better analysis of the urbanization-air quality linkage is the lack adequate air quality data for city-regions worldwide. Monitoring networks for local pollutants such as carbon monoxide, particulate matter, and nitrogen oxides are quite thin if non-existent in much of the developing world, where the majority of urban growth is likely to occur in coming decades.

Researchers have attempted to fill this gap by employing atmospheric models (typically drive by country-level emissions data), econometric models based on existing monitoring data, or more recently, satellite-derived concentration grids. The use of atmospheric models may be the most pervasive in the recent research literature, but there are drawbacks to this approach. The most pressing concern is with model resolution; most models are quite coarse both horizontally and vertically, such that it is difficult to appropriately model local concentrations within cities. Global atmospheric models are also by necessity a simplified version of reality, often leaving off important interactions such as the potential feedbacks between different pollutants, land use changes, and climate variability. Without sufficient urban-scale resolution, the models may be a poor source of data for estimating urban pollution exposure. Finer-scale regional models have been employed in some locations with more success, although few efforts have yet been made to aggregate findings to a global scale or to make comparisons across space and time.

Econometric models are equally problematic because they are built on existing monitoring data, which may be limited to certain regions of the world with particular development histories, and are often averaged over a period like one year so as to obscure important seasonal or daily variations. The new satellite-based monitoring efforts show substantial promise for obtaining global city pollution information and will likely be the foundation for future analysis of urbanization and air quality, although satellites may have a hard time resolving surface pollutants such as ozone.

Better data is also needed on the spatial distribution of pollution activity within countries, especially in the developing nations. Efforts such as EDGAR to spatially allocate national emissions to a finer grid than have been used previously show promise for evaluating urban air pollution trends and for comparing regions with similar circumstances [28]. Such datasets may be valuable to cities without adequate in-house monitoring and analysis capacity, which could use the datasets to obtain a sense of their historic and recent emissions and to identify appropriate air pollution efforts within their cities or in locations outside cities that may have an outsized impact on urban populations. In addition, national emissions scenarios such as those produced for the Intergovernmental Panel on Climate Change need to be updated and spatially-resolved at a finer scale so that cities have a better understanding of how their coming demographic and economic change may translate into pollution activity.

Better data is also needed to examine the spatial distribution of pollution within urban areas, so as to inform efforts to minimize pollution exposure. Different urban activities and land use intensities can have vastly different implications for air quality, yet the research community still has only a basic understanding of these relationships (i.e., we should move industrial activity out of the densest areas) and how to assess different land use regimes. Efforts such as the Global Land Cover Characteristics dataset should be expanded in the future and should be analyzed in more depth in a comparative context. In addition, many scholars urge further epidemiology studies on the impacts of pollution on human health, which tend to be limited to few locations that are difficult to compare.

Ultimately, these comparative data analyses need to inform policymaking; i.e., have policymakers adopted sufficient air quality management policies and controls to ensure healthy air in their communities? In places that have already adopted control policies, such as those mentioned previously, part of this effort requires documenting whether existing policy efforts are achieving their goals. Are new policies resulting in a reduction in local or regional pollution? If not, why not? What else should be done in the regions to protect human and environmental health? Answering such questions will require considerable dexterity in translating existing research with all of its gaps, caveats, and complications to actionable policy solutions.

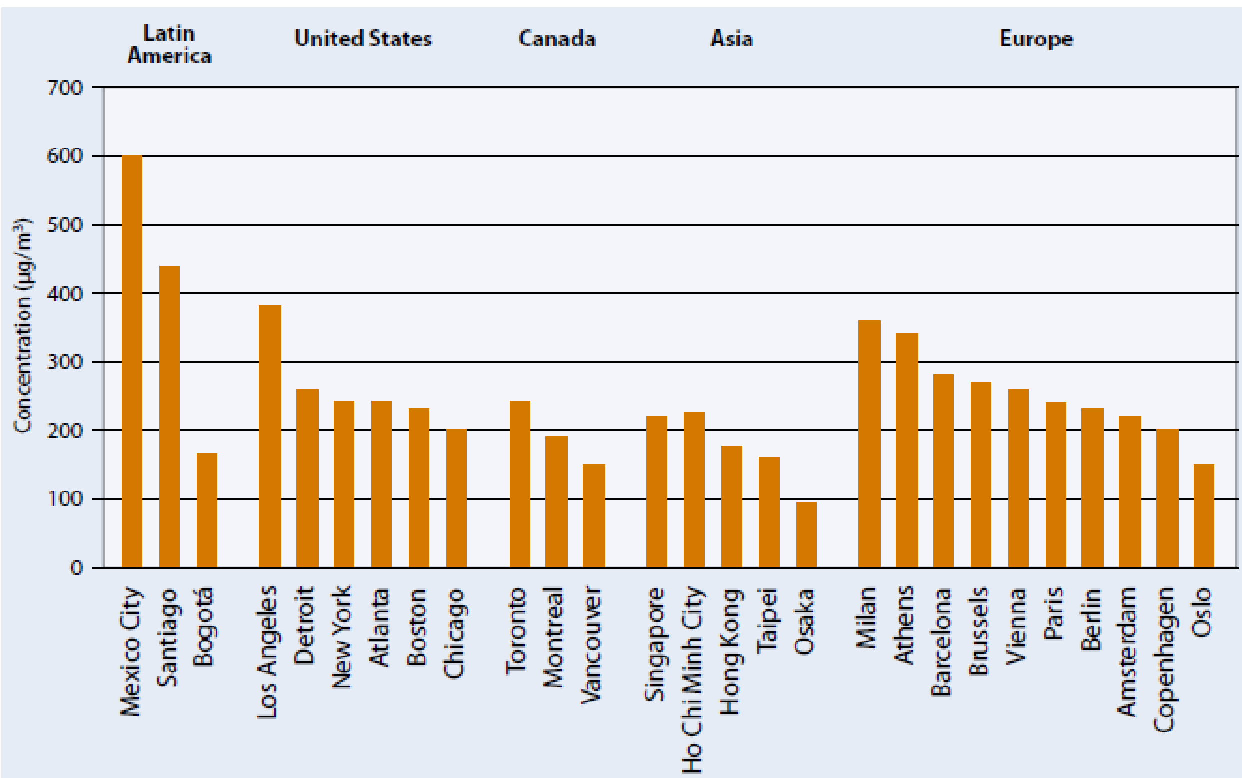
Figures and Tables

Table 1. Range of Observed Surface Pollutant Concentrations (μg/m³) based on Select Cities

Region	Annual average concentration			Ozone (1-hour maximum concentration)
	PM_{10}	Nitrogen dioxide	Sulfur dioxide	
Africa	40–150	35–65	10–100	120–300
Asia	35–220	20–75	6–65	100–250
Australia/New Zealand	28–127	11–28	3–17	120–310
Canada/United States	20–60	35–70	9–35	150–380
Europe	20–70	18–57	8–36	150–350
Latin America	30–129	30–82	40–70	200–600

Source: WHO, 2006, p.31 [13].
Note: Ozone concentrations are frequently reported in parts per billion (ppb); 1 ppb is equal to 2 μg/m³.

Figure 1. Maximum Observed Surface Ozone Concentrations (1-hr average) in Select Cities



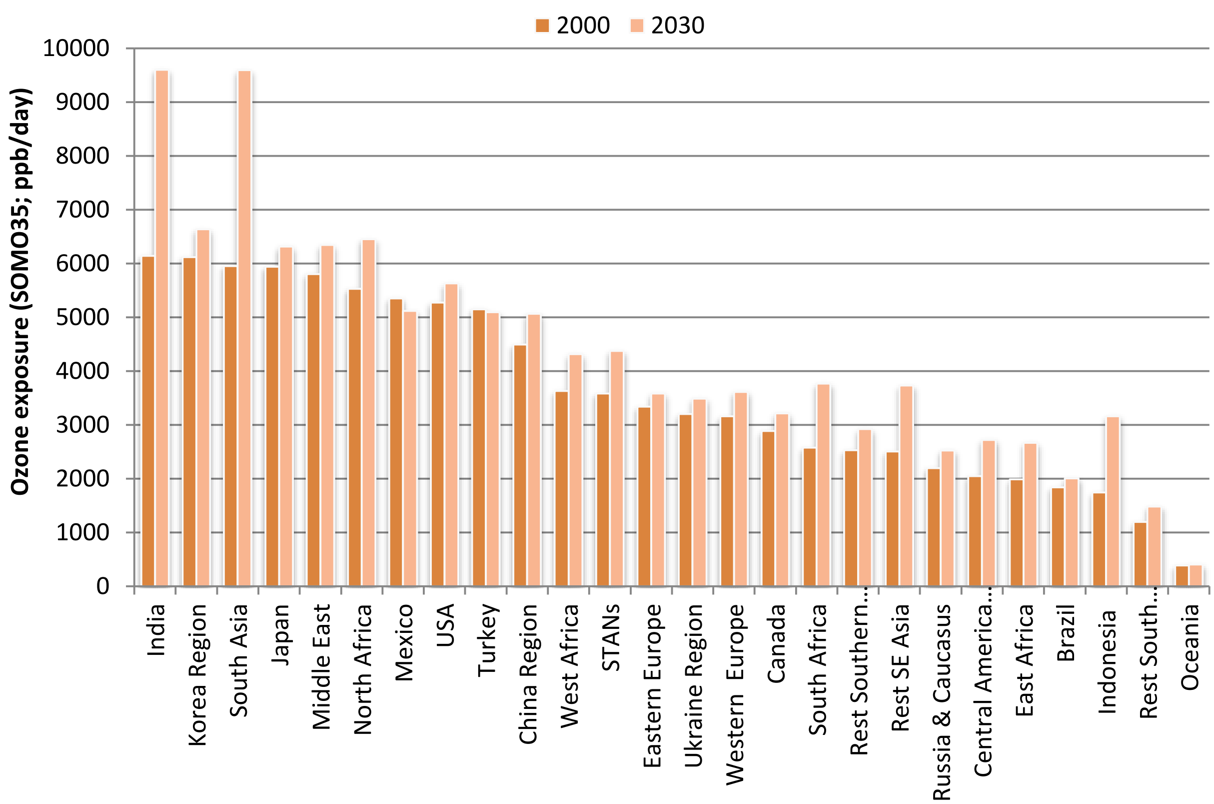
Source: WHO, 2006, p.44 [13].

Table 2. IPCC SRES Scenario Characteristics

Scenario Group	A1FI	A1B	A1T	A2	B1	B2
Population growth	low	low	low	high	low	medium
GDP growth	very high	very high	very high	medium	high	medium
Energy use	very high	very high	high	high	low	medium
Land- use changes	low-medium	low	low	medium / high	high	medium
Resource availability (conventional and unconventional oil and gas)	high	medium	medium	low	low	medium
Pace and direction of technological change	rapid	rapid	rapid	slow	medium	medium
Change favoring	fossil fuels	balanced	non-fossils	regional	efficiency & dematerialization	"dynamics as usual"

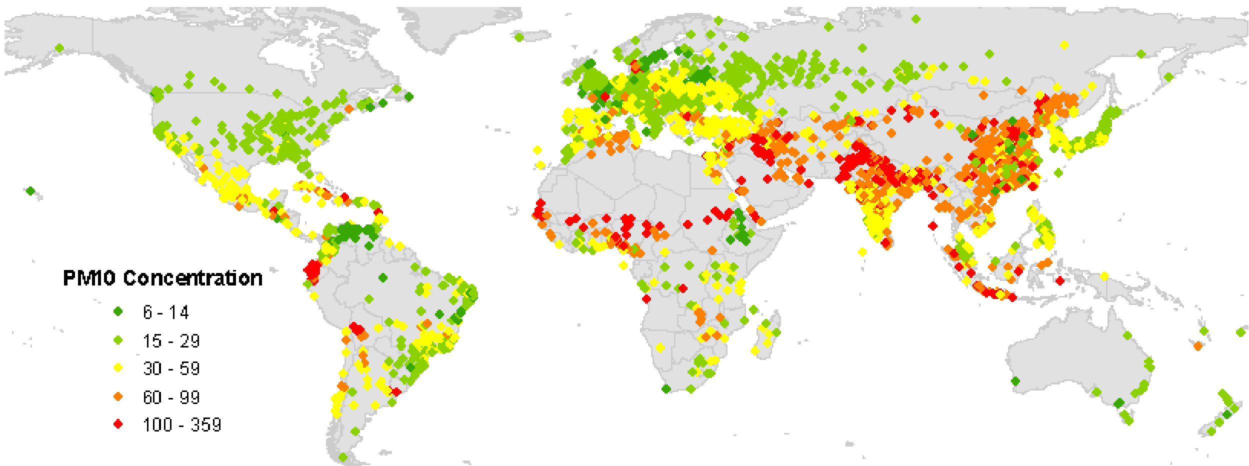
Source: IPCC SRES, 2000 [22]

Figure 2. Potential Urban Ozone Exposure, 2000 and 2030



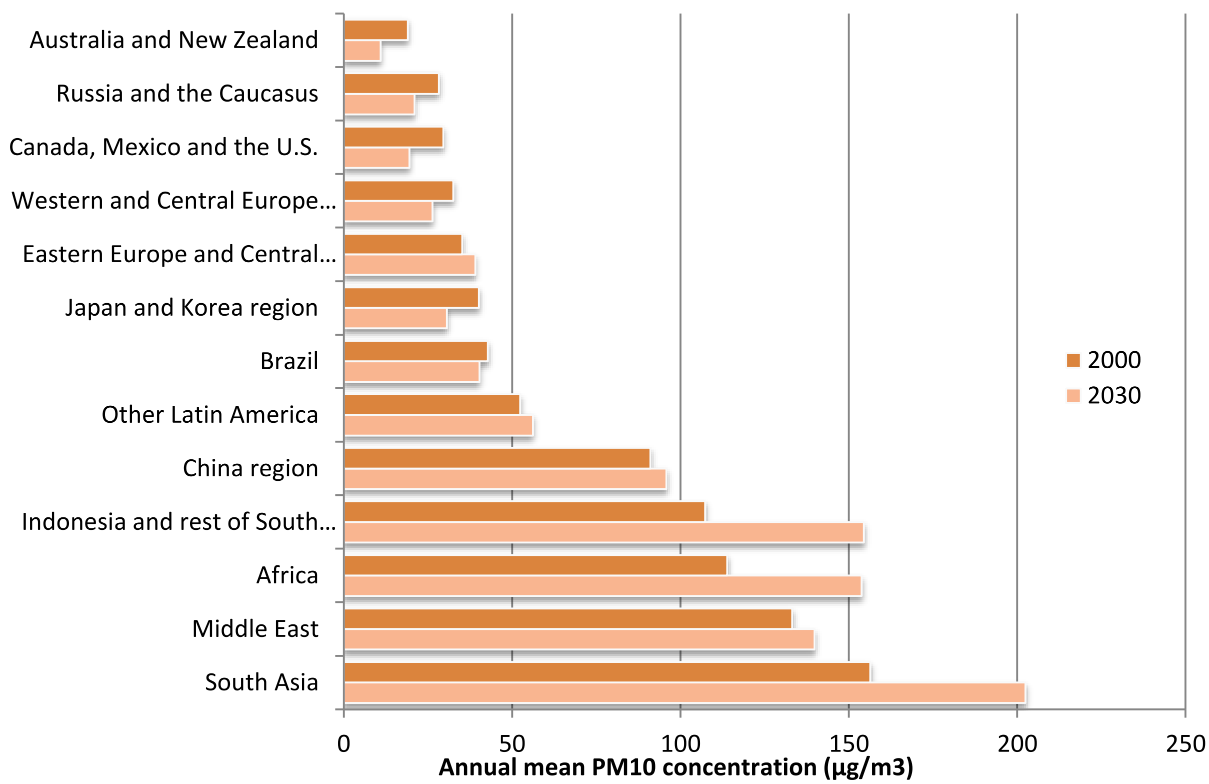
Source: OECD [10].

Figure 3. Estimated Annual Average Concentrations of PM_{10}, 2006



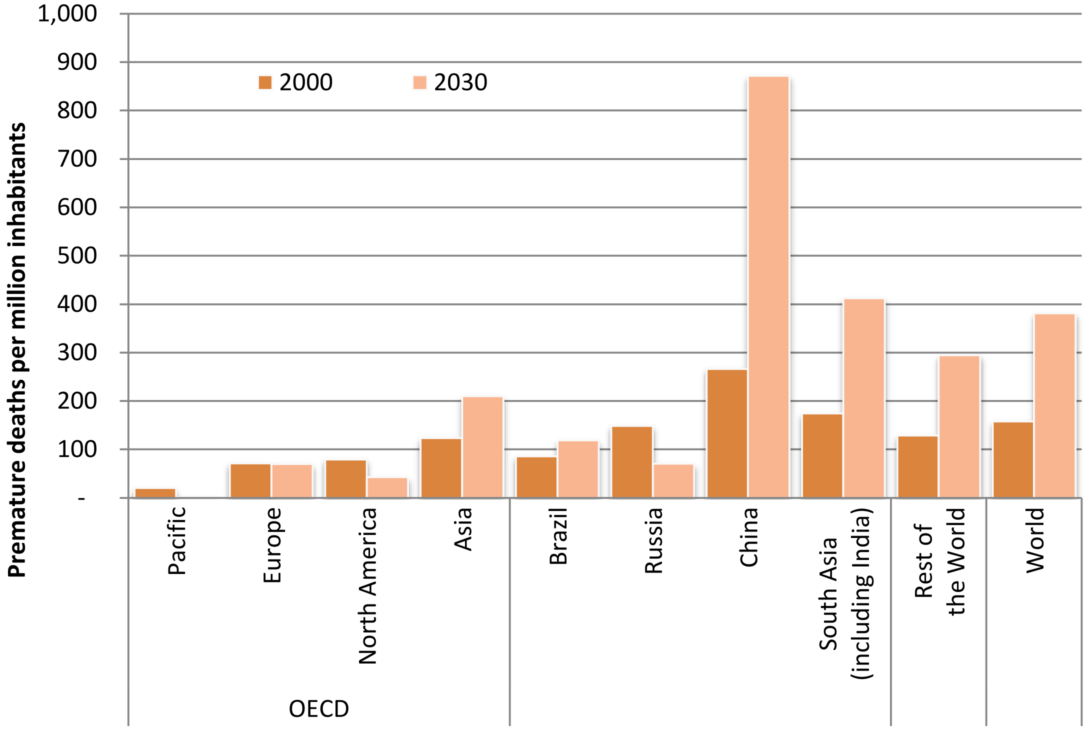
Source: World Bank Indicators

Figure 4. Estimated Annual Average Urban PM_{10} Concentrations, 2000 and 2030



Source: OECD [10]

Figure 5. Estimated Premature Deaths from Urban PM_{10} Exposure, 2000 and 2030



Source: OECD [10]

Figure 6. Sample Policy Strategies for Addressing Air Pollution in Cities

Table 1. Examples of "no-regret" actions for air pollution control in cities (compiled from delegates at the 2004 International Urban Air Quality Forum in Indianapolis, IN).

Policy initiatives	Produce and distribute unleaded gasoline
	Phase-in improved technology vehicles and engines through tighter standards
	Remove fuel subsidies
	Abolish burning of garbage and other biomass
	Lower taxes on clean products
Institutional measures	Identify and encourage champions for change
	Formulate a Clean Air Group that includes industry, fuel providers and nongovernmental organizations
Road, transport, and traffic management	Make public transportation affordable or even free for downtown destinations
	Train bus drivers about pollution and fuel use
	Promote fuel efficiency for cars and industry
	Establish one-way traffic with synchronized signals
	Pave roads, including access roads
Awareness, media, educational, and social	Publish and broadcast Air Quality Indices
	Promote a regular media outlet for air quality stories to keep up interest
	Offer environmental education in primary schools and agricultural extension services
Technical measures	Eliminate refueling leaks, establish primary VOCs recovery, as a minimum
	Reduce sulfur content of diesel fuel and gasoline to 500 ppm or lower
	Require new gasoline-powered vehicles to have operational catalytic converters
	Mandate inspection and maintenance for commercial vehicles
	Design and disseminate better stoves for coal briquettes, wood pellets, and other solid fuels
	Focus on less-polluting, better-ventilated kitchens
	Promote more efficient agricultural burning methods
Enforcement initiatives	Identify and target gross polluters
	Provide complaint phone or text message numbers for visual sighting of polluters

Source: John G. Watson [31], p.1230

REFERENCES

1. McGranahan, G., et al., *Urban Systems*, in *Ecosystems and Human Well-being: Current State and Trends, Volume 1*, R. Hassan, R. Scholes, and N. Ash, Editors. 2005, Island Press: Washington, DC. p. 795-825.
2. United Nations Population Division. *World Urbanization Prospects: The 2009 Revision.* 2010 [accessed 2010 August 17]; Data in digital form (POP/ DB/WUP/Rev.2009)]. Available from: http://esa.un.org/unpd/wup/index.htm.
3. Bolund, P. and S. Hunhammar, *Ecosystem services in urban areas.* Ecological Economics, 1999. **29**(2): p. 293-301.
4. Alberti, M., et al., *Integrating Humans into Ecology: Opportunities and Challenges for Studying Urban Ecosystems.* BioScience, 2003. **53**(12): p. 1169-1179.
5. Pickett, S.T.A., et al., *A conceptual framework for the study of human ecosystems in urban areas.* Urban Ecosystems, 1997. **1**(4): p. 185-199.
6. Scholes, R., R. Hassan, and N. Ash, *Summary: Ecosystems and Their Services around the Year 2000*, in *Ecosystems and Human Well-Being: Current State and Trends, Volume 1*, R. Hassan, R. Scholes, and N. Ash, Editors. 2005, Island Press: Washington, DC. p. 1-23.
7. Kuznets, S., *Economic growth and income inequality.* American Economic Review, 1955. **45**: p. 1-28.
8. Stern, D.I., *The Rise and Fall of the Environmental Kuznets Curve.* World Development, 2004. **32**(8): p. 1419-1439.
9. Torras, M. and J.K. Boyce, *Income, inequality, and pollution: a reassessment of the environmental Kuznets Curve.* Ecological Economics, 1998. **25**: p. 147-160.
10. Organization for Economic Cooperation and Development, *OECD Environmental Outlook to 2030*. 2008, OECD Publishing: Paris.
11. Levlieveld, J., et al., *One the role of hydroxyl radicals in the self-cleansing capacity of the troposphere.* Atmospheric Chemistry and Physics, 2004. **4**: p. 2337-2344.
12. Sillman, S., *The relation between ozone, NOx and hydrocarbons in urban and polluted rural environments.* Atmospheric Environment, 1999. **33**: p. 1821-1845.
13. World Health Organization, *Air Quality Guidelines: Global Update 2005*. 2006, Copenhagen: WHO Regional Office for Europe.
14. Horowitz, L.W., *Past, present, and future concentrations of tropospheric ozone and aerosols: Methodology, ozone evaluation, and sensitivity to aerosol wet removal.* Journal of Geophysical Research, 2006. **111**(D22): p. D22211.
15. Anenberg, S.C., et al. (2010) *An Estimate of the Global Burden of Anthropogenic Ozone and Fine Particulate Matter on Premature Human Mortality using Atmospheric Modeling*. Environmental Health Perspectives **doi:10.1289/ehp.0901220**.
16. Solberg, S., et al., *European Abatement of Surface Ozone in a Global Perspective.* Ambio, 2005. **34**(1): p. 47-53.
17. Dentener, F., et al., *The impact of air pollutant and methane emission controls on tropospheric ozone and radiative forcing: CTM calculations for the period 1990-2030.* Atmospheric Chemistry and Physics, 2005. **5**(7): p. 1731-1755.
18. Cofala, J., et al., *Scenarios of global anthropogenic emissions of air pollutants and methane until 2030.* Atmospheric Environment, 2007. **41**(38): p. 8486-8499.
19. Prather, M., et al., *Fresh air in the 21st century?* Geophys. Res. Lett., 2003. **30**(2): p. 1100.
20. Cape, J.N., *Surface ozone concentrations and ecosystem health: Past trends and a guide to future projections.* Science of the Total Environment, 2008. **400**(1-3): p. 257-269.
21. Ramaswamy, V., et al., *Radiative Forcing of Climate Change*, in *Climate Change 2001: The Scientific Basis*, J.T. Houghton, et al., Editors. 2001, Cambridge University Press: New York.

22. Nakicenovic, N., et al., *Special Report on Emissions Scenarios: A Special Report of Working Group III of the Intergovernmental Panel on Climate Change*, N. Nakicenovic and R. Swart, Editors. 2000, Cambridge University Press: Cambridge, UK.
23. Derwent, R., et al., *Global Ozone Concentrations and Regional Air Quality.* Environmental Science & Technology, 2002. **36**(19): p. 379A-382A.
24. Jaffe, D., et al., *Increasing background ozone during spring on the west coast of North America.* Geophysical Research Letters, 2003. **30**(12): p. 1613.
25. Li, Q., et al., *Transatlantic transport of pollution and its effects on surface ozone in Europe and North America.* Journal of Geophysical Research, 2002. **107**(D13): p. 4166.
26. Cohen, A.J., et al., *Urban Air Pollution*, in *Comparative Quantification of Health Risks: Global and Regional Burden of Disease Attributable to Selected Major Risk Factors*, M. Ezzati, et al., Editors. 2004, World Health Organization: Geneva. p. 1353-1433.
27. van Donkelaar, A., et al., *Global Estimates of Ambient Fine Particulate Matter Concentrations from Satellite-based Aerosol Optical Depth: Development and Application.* Environmental Health Perspectives, 2010. **118**(6): p. 847-855.
28. Sarzynski, A., *The Geography of Urban Air Pollution and Ecosystem Stress*. 2010, under development.
29. Gurjar, B.R., et al., *Evaluation of emissions and air quality in megacities.* Atmospheric Environment, 2008. **42**(7): p. 1593-1606.
30. Goyal, S.K. and C.V. Chalapati Rao, *Assessment of atmospheric assimilation potential for industrial development in an urban environment: Kochi (India).* Science of the Total Environment, 2007. **376**(1-3): p. 27-39.
31. Chow, J.C., et al., *Megacities and Atmospheric Pollution.* Journal of the Air & Waste Management Association (1995), 2004. **54**(10): p. 1226-1235.
32. Greater London Authority. *Clearing London's Air*. 2010 [accessed 2010 October 5]; Available from: http://www.london.gov.uk/clearing-londons-air.
33. Mairie de Paris. *Consequences of the Law on Air*. 2006 [accessed 2010 October 5]; Available from: http://www.paris.fr/portail/pratique/Portal.lut?page_id=6543&document_type_id=5&document_id=15987&portlet_id=14832.
34. Hong Kong Environmental Protection Department. *Incentive Scheme for Replacing Euro lI Diesel Commercial Vehicles by New Commercial Vehicles*. 2005 [accessed 2010 October 5]; Available from: http://www.epd.gov.hk/epd/english/environmentinhk/air/prob_solutions/old_diesel_com_veh_replace_prog.html.
35. Chechel, A. and Y. Vinogradova. *Transportation Tax May Be Tied to Emissions*. 2010 September 15 [accessed 2010 October 5]; Available from: http://www.themoscowtimes.com/mobile/business/article/416011.html.
36. Asian Development Bank and Clean Air Initiative for Asian Cities Center. *Country Synthesis Report on Urban Air Quality Management: Malaysia*. 2006 December [accessed 2010 October 5]; Available from: http://www.cleanairnet.org/caiasia/1412/csr/malaysia.pdf.
37. Japan for Sustainability. *Tokyo's Role in Addressing Air Pollution in Japan*. JFS Newsletter No. 31. 2005 [accessed 2010 October 5]; Available from: http://www.japanfs.org/en/mailmagazine/newsletter/pages/027794.html.
38. NYC Environmental Protection. *Air Pollution Control*. 2010 [accessed 2010 October 5]; Available from: http://www.nyc.gov/html/dep/html/air/index.shtml.
39. City of Boston. *Community Climate Action and Air Quality Grants*. 2010 [accessed 2010 October 5]; Available from: http://www.cityofboston.gov/environment/airpollution/grants.asp.

40. U.S. Environmental Protection Agency. *India Air Quality and Climate Change*. 2010 [accessed 2010 October 5]; Available from: http://www.epa.gov/international/air/india.htm.
41. U.S. Environmental Protection Agency. *The Integrated Environmental Strategies (IES) Program in Santiago, Chile*. 2005 November [accessed 2010 October 5]; Available from: http://www.epa.gov/ies/pdf/ies_chilefactsheet_final.pdf.
42. City of Austin Texas. *COA O3 Reduction Strategies*. 2010 [accessed 2010 October 5]; Available from: http://www.ci.austin.tx.us/airquality/o3red.htm.
43. City of Hong Kong. *Air Quality in Hong Kong*. 2010 [accessed 2010 October 5]; Available from: http://www.gov.hk/en/residents/environment/air/airquality.htm.
44. Gurjar, B.R., A.S. Nagpure, and T.P. Singh. *Air quality in megacities*. The Encyclopedia of Earth. 2010 April 30 [accessed 2010 October 5]; Available from: http://www.eoearth.org/article/Air_quality_in_megacities.
45. Scheer, S. *Better Place sees electric cars on roads in 2011*. 2010 February 7 [accessed 2010 October 5]; Available from: http://www.reuters.com/article/idUSTRE6160T520100207.
46. Government of Mexico City. *Do Not Drive Today*. 2010 [accessed 2010 October 5]; Available from: http://www.hoynocircula.com.mx/.
47. Molina, M.J. and L.T. Molina, *Megacities and Atmospheric Pollution.* Journal of the Air & Waste Management Association (1995), 2004. **54**(6): p. 644-680.
48. Berlin's Senate Department for Health Environment and Consumer Protection. *Better Air for Berlin*. 2007 [accessed 2010 October 5]; Available from: http://www.berlin.de/sen/umwelt/luftqualitaet/de/luftreinhalteplan/download/Umweltzone_Broschuere_en.pdf.
49. United Nations Environment Programme. *Bangkok State of the Environment 2001; Chapter 2- Air Pollution*. 2001 [accessed 2010 October 5]; Available from: http://www.rrcap.net/pub/soe/bangkok_air.pdf.
50. Beevers, S.D. and D.C. Carslaw, *The impact of congestion charging on vehicle emissions in London.* Atmospheric Environment, 2005. **39**(1): p. 1-5.
51. Pucher, J., et al., *Urban Transport Trends and Policies in China and India: Impacts of Rapid Economic Growth.* Transport Reviews: A Transnational Transdisciplinary Journal, 2007. **27**(4): p. 379 - 410.
52. Institute for Transportation & Development Policy. *Ahmedabad, India wins 2010 Sustainable Transport Award*. 2010 January 12 [accessed 2010 October 5]; Available from: http://www.itdp.org/index.php/projects/update/ahmedabad_2010_sustainable_transport_award/.
53. South Coast Air Quality Management District. *Over a Dozen Years of RECLAIM Implementation: Key Lessons Learned in California's First Air Pollution Cap-and-Trade Program*. 2007 [accessed 2010 October 5]; Available from: http://www.aqmd.gov/reclaim/docs/Policy_Paper_Part1.pdf.
54. Schreifels, J. *Emissions Trading in Santiago, Chile: A Review of the Emission Offset Program of Supreme Decree No 4*. [accessed 2010 October 5]; Available from: http://www.epa.gov/airmarkets/international/chile/et_santiago.pdf.
55. Bay Area Air Quality Management District. *Particulate Matter*. 2010 [accessed 2010 October 5]; Available from: http://www.baaqmd.gov/Divisions/Planning-and-Research/Particulate-Matter.aspx#erParticulate.
56. South Coast Air Quality Management District. *AQMD Adopts Measure to Reduce Dust and Particulate Emissions from Wide Range of Industries*. 2009 [accessed 2010 October 5]; Available from: http://www.aqmd.gov/NEWS1/2009/bs_12_04_09.htm.
57. Hsu, D., *Sustainable New York City*. 2006, Design Trust for Public Space and the New York City Office of Environmental Coordination: New York.